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**Remedial Investigation Report
for the
Eastern Michaud Flats Site**

**Part II
Surface and Subsurface Characterizations**

**Volume IV
Sections 4.5 – 5**

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4.5 SURFACE WATER AND SEDIMENTS

This section provides an assessment of the nature and extent of EMF-related constituents in surface water, sediments, and springs associated with the river. The assessment is based on the results of the surface water and sediment sampling performed as part of the RI. The sampling and analysis program for the surface water and sediment investigation was described in Section 2.4 of this report.

Phase I consisted of sampling at locations ranging from the City of Pocatello, approximately 6 miles (9.6 km) upstream of the EMF facilities, to River Mile 10, approximately 4 miles (6.4 km) downstream of the EMF facilities (Figures 4.5-1 and 4.5-1a to g). Water samples were collected from 27 locations during four events: July 1992, October 1992, February 1993, and April 1993; sediments were collected in July 1992. Flow gaging in the Portneuf River was performed at selected locations during these sampling events. Samples collected at nine of the 27 locations were spring samples as opposed to river water samples, and thus reflected groundwater chemistry. The springs included Batiste and Swanson Road Springs also sampled as part of the groundwater monitoring program discussed in Section 4.4.

Phase II consisted of surface water and sediment sampling at locations in the immediate vicinity of the FMC IWW ditch outfall (Figure 4.5-1h) and sediment sampling in the Fort Hall Bottoms (approximately 5.5 miles [8.8 km] downstream from the EMF facilities) (Figure 4.5-1i). All Phase I and II samples were analyzed for a suite of metals, nutrients, common ions, fluoride and radiological parameters.

Because the EMF facilities have been in operation for more than 40 years, it was assumed that cumulative effects of chemicals transported to the river from the EMF facilities would be evident in sediments collected along the Portneuf River near the EMF facilities. When measureable impact on sediment proved limited to the immediate area of the FMC IWW ditch outfall, and there was no measurable impact on surface water, another investigation was initiated at EPA's request at the confluence of the Portneuf River and the American Falls Reservoir. Results of this investigation are presented in Section 4.6, Ecology.

Surface Water and Sediment Investigation Objectives

The objectives of the surface water and sediment investigation were:

- To assess the nature and extent of any EMF-related impacts on the Portneuf River water and sediments.
- To evaluate the pathways by which chemicals originating from the EMF facilities may be transported to the river. The four potential pathways are:
 - Direct aerial deposition,
 - Surface runoff from impacted surface soils,
 - Discharge of impacted groundwater
 - Direct discharge (i.e., the IWW ditch outfall)

Overview of Findings

The major findings of the surface water and sediment investigation are listed below. Data evaluation methods used to arrive at these findings included comparisons of upstream and downstream results; comparison of results with soil and groundwater representative levels; and application of various statistical techniques, including cluster analyses, t-tests, and non-parametric ANOVAs.

- There were no measureable effects on surface water chemistry directly attributable to the EMF facilities. Surface water upstream from the EMF facilities contained lower sulfate, nitrate, and total phosphorus concentrations than river water downstream of the facilities; however, this result is explained by the high rate of groundwater unaffected by the EMF facilities discharging to the river (200 cfs between the EMF facilities and Siphon Road). In addition, there are other documented sources of nitrate, sulfate, and total phosphorus to the Portneuf River downstream from the EMF facilities.
- EMF effects on sediments were limited to samples SD17 and SD17A, collected at the IWW ditch outfall.
- Because there were no measurable effects on sediment chemistry attributable to the EMF facilities beyond the localized area of the IWW ditch outfall, aerial deposition and surface soil runoff are not significant transport pathways to surface water and sediment. This

conclusion is further supported by results for specific samples most likely to reflect the influences of these pathways (sediment samples SD9 and SD11).

- Consistent with Section 4.4 findings, groundwater discharging at Batiste and Swanson Road Springs contained EMF-related constituents. Arsenic, barium, boron, and lithium, and ammonia, nitrate, total phosphorus, and sulfate exceeded representative groundwater levels in one or more samples from these springs. However, the average concentrations of these chemicals at these springs were not significantly above representative groundwater levels. In fact, average concentrations of arsenic and nitrate were below representative groundwater levels. None of these constituents were identified at elevated levels in samples collected immediately downstream of Batiste or Swanson Road Spring.
- Constituent concentrations were not elevated in river water at the IWW ditch outfall. (A comparison of data for groundwater from FMC production well FMC-1, the source of the non-contact cooling water discharged to the IWW ditch; water from the IWW ditch; and surface water collected at the IWW ditch outfall is presented in Tables 4.5-1 and 4.5-1a).

Section Content and Organization

An overview of the organization and conclusions of Section 4.5 is provided on Figure 4.5-2. The results, data evaluation methods, and findings of the surface water investigation are presented in Section 4.5.1. Section 4.5.2 presents the results, data evaluation methods, and findings for sediment.

RI surface water and sediment sampling results are presented in Appendix U.

<p>CHARACTERIZATION OF SURFACE WATER (SECTION 4.5.1)</p>	<p>Overall Discussion (Section 4.5.1.1)</p> <ul style="list-style-type: none"> • There do not appear to be any representative level exceedances downstream directly attributable to the EMF facilities despite above-representative levels of EMF-related constituents detected at Batiste and Swanson Road Springs. 	<p>Statistical Methods (Section 4.5.1.2)</p> <ul style="list-style-type: none"> • The springs can be divided into groups based on spring water chemistry. Batiste and Swanson Road Spring chemistry are unique. • The general chemistry of the groundwater discharging to the river is different from that of the upstream river water. As expected, downstream river water is more similar to groundwater than to the upstream water under low-flow conditions. • Mixing zone effects were generally not apparent downstream of the EMF facilities. 	<p>Detailed Discussion (Section 4.5.1.3)</p> <ul style="list-style-type: none"> • This section provides a chemical-by-chemical comparison of down-stream river water with groundwater and upstream river water.
<p>CHARACTERIZATION OF SEDIMENTS (SECTION 4.5.2)</p>	<p>Overall Discussion (Section 4.5.2.1)</p> <ul style="list-style-type: none"> • The only sediment samples which reflect EMF influences are SD17 and SD17A collected at the FMC IWW outfall. • Above-representative level constituent concentrations were not detected in downstream samples 	<p>Statistical Methods (Section 4.5.2.2)</p> <ul style="list-style-type: none"> • With few exceptions, near-site, spring and downstream sediment constituent concentrations were not statistically different from upstream concentrations. Constituents for which statistical differences were found were often higher upstream from the EMF facilities than they were downstream. • Samples from the IWW outfall (SD17 and SD17A) were very different from all other samples, underscoring the conclusion that SD17 reflects IWW ditch influence and that measurable effects of this influence are localized at the outfall. 	<p>Detailed Discussion (Section 4.5.2.3, River Sediments and Section 4.5.2.4, Spring Sediments)</p> <ul style="list-style-type: none"> • Upstream sediment constituent concentrations were very similar to soil representative levels. • These sections provide a sample-by-sample discussion of the river and spring sediment sample analytical data. • Aerial deposition and surface water runoff do not appear to be significant transport pathways as evidenced by the results for sediment samples SD9 and SD11.

**FIGURE 4.5-2
OVERVIEW OF SURFACE WATER AND SEDIMENT INVESTIGATION**

4.5.1 NATURE AND EXTENT OF EMF-RELATED CONSTITUENTS IN SURFACE WATER

The nature and extent of EMF-related constituents in surface water were investigated by two methods. The first was a multivariate statistical method called cluster analysis. This was performed to assess the degree of dissimilarity of samples collected beyond the potential influence of the EMF facilities with those collected downstream. The second was a chemical-by-chemical comparison of constituent concentrations with representative groundwater concentrations and upstream surface water concentrations. During low-flow conditions along the Portneuf River, the comparison of downstream surface water samples with groundwater representative levels is valid due to the relatively large volume of groundwater discharged to the river downstream from the EMF facilities. Three sampling events occurred during low-flow conditions (less than one-half average flow), and one event occurred during above-average flow conditions. For the purposes of this investigation, gaining reach river water quality was compared to background groundwater chemistries as defined in Section 4.4, since groundwater from all three hydrogeochemical regimes discharges to the river.

The results of these comparisons and analyses were used along with the understanding of surface water hydrology presented in Section 3.3 and knowledge of EMF and non-EMF potential sources to draw conclusions as to the nature and extent of EMF effects on surface water.

4.5.1.1 Surface Water Chemistry Data – Overall Results

The following discussion of surface water chemistry within the EMF study area draws on the data summarized in Tables 4.5-2 through 4.5-8. Appendix U presents metals analysis results with validation qualifiers for individual samples collected during each round of RI sampling.

Antimony, beryllium, cadmium, chromium, cobalt, lead, molybdenum, nickel, selenium, silver, thallium, and zinc were either not detected in any of the water samples or were detected only in concentrations at their detection limits. These constituents are not discussed further in this section. Mercury was reported by the laboratory to be present in several surface water samples at

levels just above its detection level. However, these results are considered to be false-positives as discussed in Section 4.1.

Total aluminum, iron, and manganese were detected above representative groundwater levels at a number of locations. However, their concentrations correlate well with turbidity and river discharge, and are thus likely indicative of naturally occurring suspended solids in the river system. Nevertheless, these elements are also discussed in Section 4.5.1.3.

Arsenic, ammonia, barium, boron, fluoride, lithium, nitrate, phosphorus, and sulfate were found at concentrations above representative groundwater levels at Batiste Spring (SW14) and Swanson Road Spring (SW15). Although some of these chemicals exceeded representative groundwater levels in one or more downstream samples, the exceedances do not appear to be attributable to EMF. A detailed discussion of the above-listed constituents is provided in Section 4.5.1.3.

Copper was detected at mean concentrations in excess of the representative groundwater concentrations at the IWW ditch outfall (0.015 mg/l total copper), but these levels did not exceed the mean concentrations for upstream sampling station SW19.

4.5.1.2 Surface Water Statistical Analyses – Methods and Results

Data presented in Tables 4.5-2 through 4.5-7 are mean concentrations of analytes. The mean concentrations at each sampling station were calculated using results from four samples collected over a one year period, when available. Constituents reported as not detected were not used in the calculation of mean concentrations. Omission of the nondetects when calculating mean concentrations is considered a conservative approach because it typically leads to higher mean concentrations for comparison with the representative groundwater concentrations, which were calculated using the detection limit values. This approach exaggerates surface water concentrations with respect to groundwater concentrations.

A multivariate statistical analysis called cluster analysis was used to investigate the possibility that surface water samples collected within the channel of the Portneuf River were within the "mixing zone" of nearby discharges (e.g., STP), and thus were not representative of ambient Portneuf River (Park, 1974). Cluster analysis was also used to investigate groupings or clusters within the dataset that are not immediately evident by inspection. Cluster analysis is used for investigating patterns in datasets using multiple variables concurrently.

For this analysis, constituents displaying the highest degree of dissimilarity were used. These were: calcium, arsenic, barium, bicarbonate, fluoride, potassium, lithium, magnesium, sodium, ammonia, nitrate, orthophosphate, total phosphorus, and sulfate. This group of constituents includes those transported via various pathways. Use of these variables increased the overall contrast between samples or sample "clusters".

The cluster analysis confirmed that certain springs form distinct groups. Samples SW13 (STP), SW9 (FMC Employee Park), SW15 (Swanson Road Spring), and SW14 (Batiste Spring) all define separate clusters, indicating unique chemistry associated with each spring. Springs located further north (SW2, SW5, SW7, SW6, and SW4) are similar to each other and dissimilar from the other springs. SW9 is more similar to the northern springs, and less similar to the springs near the EMF facilities and the STP.

This analysis also indicates that spring chemistry is distinct from the river chemistry, regardless of season or river discharge. Samples from SW11, in the spring drainage downstream of Batiste Spring, are more similar to river samples than the Batiste Spring samples. In other words, EMF-related influences detected at Batiste Spring are no longer apparent in the surface water along the spring drainage several hundred feet downstream. This finding is not unexpected, because the drainage channel from Batiste Spring triples in flow rate between the spring house (SW14) and the point at which it meets the main river channel, providing ample water to dilute the EMF-influenced water discharged at Batiste Spring.

For some sampling events, SW11 is more similar to SW10 or SW12, both downstream river sampling points. This indicates that the gaining river water is more similar to representative groundwater than the upstream river water. This finding is expected because the river gains more than 200 cfs from groundwater discharge, and during low-flow conditions, upstream river flow is only 20 to 150 cfs. In general, samples from unaffected springs are similar to the gaining reach river water during low flow events. This provides further support to the conclusion that gaining reach river water is more similar to groundwater chemistry than it is to upstream surface water chemistry. These results mean that, under low-flow conditions, comparing downstream surface water chemistry with background groundwater chemistry is a valid means of assessing potential EMF-related influences on surface water quality in the river.

Samples from the upstream river reach form four distinct groups, one for each sampling event. This clustering indicates that the upstream water chemistry is fairly consistent throughout the losing river reach, up to station SW16. SW17, near the FMC IWW ditch outfall, is not similar to any other river or spring samples, but the SW17 samples are not similar to one another, indicating temporal variation. The sample collected during April 1993 at SW25, furthest upstream from the EMF facilities, is markedly different from samples collected further downstream. This difference indicates there may have been a point source impact at SW25 during this sampling event, but there is not a measurable impact further downstream.

The April 1993 results are unique along the entire river reach in that the upstream samples (excluding SW25) and downstream samples are more similar to each other than the upstream versus downstream samples from low river discharge sampling events. This is expected because the river had very high flows during the April 1993 sampling event, and any influences from groundwater along the gaining reach will be lessened by high river flow associated with regional surface water runoff from snow melt and spring rains.

Mixing Zone Effects. The cluster analysis supports the conclusion (1) that certain sample locations were subject to mixing zone effects, and (2) that mixing zone effects were not prevalent throughout the year nor were these effects dominant in the overall sample network. To

illustrate, one prediction is that the water samples collected at a location influenced by "mixing zone effects" would be similar to the point discharge water chemistry, and unlike the upstream river water. Another prediction is that the water collected from a mixing zone should have a distinct water chemistry, especially if the point discharge water chemistry is distinctly different from river water chemistry. However, such predictions were not borne out by the data except under low flow conditions at one sampling location (SW5).

During low flow periods, there appears to have been a "mixing zone effect" observed at SW5, located in the river channel downstream from the Papoose Springs Fish Farm. During low flow periods, SW5 samples were more similar to the Papoose Spring samples SW7 and SW6. During high river flow in April 1993, SW5 was more similar to river stations SW1 and SW3A. These results indicate that SW5 is more representative of the Papoose Spring water than river water during low flow conditions. However, during higher flow, SW5 is more representative of river water. Mixing zone effects were not as obvious at SW10, located downstream from the outfall of Batiste Springs, or at SW12, at the STP outfall. In fact, SW10 was not similar to either Batiste Spring sampling location SW11 and SW14, but in several instances, the SW10 samples were most closely linked to SW12, near the STP outfall. This pattern indicates, that if there is a water chemistry signature from the STP discharge, it is observable at SW10.

Mixing zone effects were only observed at SW5 under low flow conditions in the river; the constituents discharging from Papoose Spring and influencing SW5 under these conditions are not associated with the EMF facility. It was demonstrated in Section 3.3 that groundwater from the EMF facilities does not flow toward Papoose Spring. With these preceding exceptions, the samples collected within the Portneuf River are, consequently, not biased by influences from nearby point source discharges. Thus, they adequately document ambient water quality within the river at the time of sampling.

4.5.1.3 Surface Water – Detailed Discussion

A detailed discussion of the surface water sampling results with particular focus on those constituents that exceeded upstream mean concentrations or the representative groundwater

concentrations is provided below. The discussion provides additional support for conclusions presented in Sections 4.5.1.1 and 4.5.1.2 about the nature and extent of EMF-related constituents in surface water.

This section focuses on the constituents detected at elevated concentrations in Batiste Spring and Swanson Road Spring (i.e., those constituents known to be transported from the EMF source areas to surface waters). These constituents include ammonia, arsenic, barium, boron, fluoride, nitrate, lithium, total phosphorus, and sulfate. Copper was detected above groundwater representative levels at the IWW ditch outfall sampling point in the river, and is included in the detailed discussion of river sampling results. Vanadium is discussed because results from the July 1992 sampling event appear to be affected by laboratory or field artifacts, not because vanadium was detected at elevated concentrations in the groundwater pathway or the IWW ditch discharge. Aluminum, iron, and manganese in the river samples are also discussed, although these constituents correlate with turbidity and river discharge and are not believed to be associated with the EMF facilities.

Metals

Arsenic in Springs. Arsenic was detected in at least two rounds of sampling for all spring and spring-drainage sampling points. Highest mean arsenic concentrations were at Batiste Spring (0.032 mg/l dissolved) and Swanson Road Spring (0.010 mg/l dissolved) (Table 4.5-2). These mean concentrations were higher than or equal to the representative concentrations for groundwater related to the discharges at Batiste Spring and Swanson Road Spring (0.018 mg/l Bannock Range regime associated with Batiste Spring, and 0.0104 mg/l Portneuf River Valley regime associated with Swanson Road Spring). The highest mean arsenic concentrations for the East Side System and Papoose System springs and spring-drainage sampling points were below representative groundwater levels.

The maximum arsenic concentration at Swanson Road Spring (0.0134 mg/l dissolved) occurred during the October 1992 sampling event. The maximum arsenic concentrations for Batiste

Spring (0.057 mg/l dissolved; 0.032 mg/l total) occurred during the April 1993 sampling event. However, the dissolved arsenic concentration is questionable and likely biased-high because it was greater than the total arsenic concentration.

At sampling point SW11, arsenic was detected during only two events. The maximum concentration (0.008 mg/l total) was detected during April 1993.

Arsenic in River. Arsenic concentrations in river water were low compared with concentrations in representative groundwater (Table 4.5-3). Arsenic was detected in at least two rounds of sampling for all river sampling stations except SW16 (and SW18, which was only sampled once). Mean total arsenic concentrations were marginally higher in the losing-reach group of river sampling stations (0.006 mg/l) than in the gaining-reach group (0.004 mg/l). Highest individual station means were found in the four losing-reach stations, SW20 through SW18 plus SW16. These four river sampling stations, along with SW17, are nearest to and downstream of the EMF facilities. However, the means calculated for these sampling points are based on two samples rather than the four taken. The two samples not used were below detection limits or rejected in the validation process. If mean concentrations were calculated using all four samples, the mean arsenic concentrations would have been considerably lower at these four sampling stations. In the gaining reach, arsenic concentrations in river water were comparable to concentrations in representative groundwater.

Barium in Springs. Barium concentrations in springs were comparable to representative groundwater levels. Barium was detected routinely in samples from spring and spring-drainage sampling points. The mean barium concentrations ranged from 0.064 to 0.123 mg/l, which are less than the representative groundwater levels for all sampling points except Twenty Springs-East (SW02). The mean total barium concentration at Twenty Springs-East is 0.760 mg/l. This mean concentration may have been artificially high because of a single measurement (2.81 mg/l during July 1992). Using only subsequent sampling data to calculate the mean total barium

concentration for SW02 yields a value of 0.077 mg/l. This lower mean concentration is consistent with mean concentrations for other springs.

Barium in River. Barium was detected routinely in samples from all river sampling stations. However, all mean barium concentrations were below the representative levels for groundwater (0.12 mg/l, Bannock Range; and 0.17 mg/l, Portneuf River Valley). The widespread distribution of this parameter suggests that barium is naturally occurring in river water.

Boron in Springs. Boron was detected in at least two rounds of sampling for all spring and spring-drainage sampling points. Highest mean boron concentrations were at SW15, Swanson Road Spring (0.28 mg/l total and 0.21 mg/l dissolved), and SW13, the springs near the STP (0.24 mg/l total and 0.22 mg/l dissolved). However, these concentrations are near or below the representative groundwater levels for the Portneuf River Valley hydrogeochemical regime (0.25 mg/l). In addition, Batiste Spring (SW14), Batiste Springs drainage (SW11), and Papoose Spring (SW07) also had mean boron concentrations below representative levels (0.308 mg/l, Bannock Range). Since boron was found in all four spring groups at similar levels and only two springs discharge groundwater affected by EMF-related activities, the boron was most likely naturally occurring at the levels noted above.

Boron in River. Boron was detected in at least two rounds of sampling at all river sampling stations. The highest mean boron concentrations were detected in the Phase I samples at SW17 (0.38 mg/l total and 0.23 mg/l dissolved) (Table 4.5-1). At sampling stations SW25, SW24, SW23, SW19, SW16, SW12, and SW10 mean total boron concentrations ranged from 0.27 to 0.33 mg/l, compared with the representative levels for groundwater of 0.31 mg/l for Bannock Range and 0.25 mg/l for Portneuf River Valley (Table 4.5-3). The maximum boron concentration detected during subsequent sampling was 0.11 mg/l. In general, the boron detected in the river samples was not elevated downstream from EMF discharges.

Copper in River. Mean copper concentrations in Table 4.5-3 typically represent one or two samples at each station in which copper was reported. As discussed in Section 4.4, the groundwater pathway is not transporting copper to surface waters, nor is copper an EMF-related constituent at source areas.

For river sampling station SW17 at the FMC IWW ditch outfall, mean Phase I copper concentrations (0.015 mg/l total and 0.011 mg/l dissolved) were approximately two times the mean concentration for all river samples (0.007 mg/l) for both total and dissolved copper. The Phase II sampling data at SW17 had a mean copper concentration of 0.007 mg/l, with values ranging from ND to 0.011 mg/l. The Phase I SW17 results indicate that the IWW ditch was transporting groundwater containing representative levels of copper. Additionally, there was a higher copper concentration detected at an upstream station (0.022 mg/l at SW19), indicating copper concentrations in surface are variable.

Lithium in Springs. Lithium was detected in at least three rounds of sampling of all spring and spring-drainage sampling points. The highest mean lithium concentrations were at SW14, Batiste Spring (0.051 mg/l total and 0.053 mg/l dissolved; Table 4.5-2). These concentrations were above the representative level for Bannock Range groundwater (0.0165 mg/l).

Lithium concentrations for the Papoose Spring system (SW05, SW06, SW07) ranged from not detected to 0.038 mg/l (total) and from 0.024 to 0.039 mg/l (dissolved), greater than the representative level for Bannock Range groundwater (0.0165 mg/l), but less than the Michaud Flats and Portneuf River Valley representative levels (0.040 and 0.061 mg/l). Mean lithium concentrations for Swanson Road Spring (SW15) and East Side springs (SW09 and SW13) ranged from 0.023 to 0.044 mg/l, and were comparable to the representative level (0.040 mg/l) for Portneuf River Valley representative groundwater.

Lithium levels were higher in river water upstream of the EMF operations areas (SW23 to SW25) than in springs (Tables 4.5-2 and 4.5-3). Figure 4.5-5 illustrates this trend in lithium concentrations for spring sampling points and for river sampling stations.

Lithium concentrations for other springs in river water upstream of EMF facilities were also higher than the representative level for groundwater (0.0165 mg/l, Bannock Range; and 0.040 mg/l, Portneuf River Valley), but they most likely represent naturally occurring levels, similar to higher lithium concentrations.

Lithium in River. Lithium was detected in samples from all river sampling stations except gaining-reach stations SW12E (dissolved lithium) and SW7E (total lithium). Mean lithium concentrations for all river sampling stations were comparable to or higher than the representative levels for groundwater. Upstream from the EMF facilities, lithium was present at higher levels in river water than in representative groundwater, and its presence does not represent an impact from the EMF facilities.

As shown in Table 4.5-3, mean lithium concentrations in samples from river sampling stations decreased from a high value of 0.058 mg/l total lithium at SW25 and SW24 to a mean concentration of 0.037 mg/l (total and dissolved) for lithium in the gaining-reach sampling stations. Figure 4.5-3 illustrates this trend in lithium concentrations.

Vanadium in Springs. Vanadium concentrations were near detection limits in most samples from spring and spring-drainage sampling points in the EMF study area. These concentrations were below the representative levels for groundwater (0.10 mg/l, Bannock Range; and 0.199 mg/l, Portneuf River Valley).

The mean vanadium concentrations presented in Table 4.5-2 are not a clear representation of vanadium detected over four rounds of surface water sampling, as concentrations varied by two orders of magnitude. During the initial round of surface water sampling in July 1992, reported vanadium concentrations for spring-related sampling points (0.04 to 0.13 mg/l) were much

higher than those reported for subsequent rounds of sampling (maximum 0.011 mg/l). During the April 1993 round of sampling, vanadium was not detected in any surface water sample.

Vanadium concentrations are illustrated for a sampling point from each of the four spring systems in Figure 4.5-4. The higher vanadium concentrations detected among the samples collected during the July 1992 sampling event may reflect the influence of field or laboratory procedures which resulted in artificially high vanadium concentrations.

Vanadium in River. Vanadium concentrations were near detection limits in samples from river sampling stations in the EMF study area. There was a small increase in the mean vanadium concentrations from upstream to downstream; however, the vanadium concentrations in gaining reach river water were below the representative levels for groundwater (0.100 mg/l, Bannock Range; and 0.199 mg/l, Portneuf River Valley).

The mean vanadium concentrations presented in Table 4.5-3 are not a clear representation of vanadium detected over four rounds of surface water sampling, as concentrations varied by an order of magnitude. During the July 1992 sampling event, the reported vanadium concentrations for six river sampling stations were much higher (0.04 to 0.08 mg/l) than for subsequent rounds of sampling (maximum 0.003 mg/l). Vanadium was reported as "not detected" for the remaining nine river sampling stations during the July 1992 sampling event with detection limits ranging from 0.015 to 0.190 mg/l. During the April 1993 round of sampling, vanadium was not detected in any surface water samples, and sample detection limits were 0.004 mg/l.

The vanadium concentrations for river sampling stations are illustrated in Figure 4.5-5. Based on four rounds of sampling, it is possible that these concentrations are associated with a seasonal fluctuation in concentrations. However, the "trend" is more likely an effect of field or laboratory procedures which resulted in artificially high vanadium concentrations for July 1992.

Aluminum in River. Total aluminum was detected routinely in samples from the majority of river sampling stations in the losing reach of the Portneuf River: SW25, SW24, SW23, SW20, SW19, and SW16. Aluminum concentrations in samples from SW16 and SW25

are illustrated in Figure 4.5-6. At other sampling locations, total aluminum (Table 4.5-3) was detected in only one or two samples. Dissolved aluminum (Table 4.5-3) was detected in only one or two samples collected from each location.

The presence of aluminum in surface water samples as total aluminum rather than dissolved aluminum is generally an indicator of a turbid water sample due to the presence of suspended solids. Total aluminum was detected in all river samples for April 1993 when riverflow was at a maximum for all sampling events.

Iron in River. Total iron was detected in river water samples as a result of suspended solids. Seasonally high levels of total iron resulted from increased turbidity that occurred during periods of increased flow in the Portneuf River. Total iron was routinely detected at all river sampling stations except SW25 and was present at higher concentrations in the losing reach than in the gaining reach of the Portneuf River.

Mean dissolved iron concentrations (Table 4.5-3) for all the river sampling stations and mean total iron for SW25 reflect only one or two samples in which iron was reported. Dissolved iron was near detection limits in all river water samples from the EMF study area.

The mean total iron concentrations are not a clear representation of iron detected over four rounds of surface water sampling as concentrations varied by two orders of magnitude. Total iron concentrations ranged from below detection to 0.32 mg/l for all river samples for the first three rounds of sampling (Appendix U). However, total iron concentrations ranged from 0.94 to 1.73 mg/l in river water samples during the April 1993 sampling event. Figure 4.5-7 illustrates this trend in total iron concentrations for SW22 and SW16 in the losing reach, and SW10 and SW08 in the gaining reach.

Comparison of group means provided in Table 4.5-3 for losing-reach versus gaining-reach river stations shows that during both the low flow (first three events) and high flow (April 1993) sampling events, iron concentration were greater in the losing-reach than the gaining-reach.

Manganese in River. Total manganese was routinely detected at all river sampling stations except SW21, and was present at higher concentrations in the losing reach than in the gaining reach of the Portneuf River. For the July 1992 round of sampling, manganese was reported in only two samples (0.037 mg/l for SW20 and 0.012 mg/l for SW01).

Total manganese was detected in river water samples due to the presence of suspended solids; manganese was not present in filtered river water samples. Seasonally high levels of total manganese resulted from increased turbidity which occurred during periods of increased flow in the Portneuf River.

Similar to aluminum and iron, the presence of manganese in surface water samples as total manganese rather than dissolved manganese was generally an indicator of a turbid water sample. The conclusion drawn from this observation is further supported by comparing total manganese concentrations with river flow. Total manganese was detected in all river samples for April 1993, when river flow was at a maximum for all sampling events.

The mean total manganese concentrations presented in Table 4.5-3 are not a clear representation of manganese detected over four rounds of surface water sampling, as concentrations increased twofold to fourfold for the April 1993 sampling event. Total manganese concentrations ranged from below detection to 0.014 mg/l. However, total manganese concentrations ranged from 0.037 to 0.062 mg/l during the April 1993 sampling event. The total manganese concentrations for sampling stations SW25, SW16, SW12, and SW03 are illustrated in Figure 4.5-8.

Comparison of group means provided in Table 4.5-3 for losing-reach versus gaining-reach river sampling stations shows that total manganese concentrations were approximately the same for the losing-reach and gaining-reach river sampling stations.

Nutrients, Fluoride, and Sulfate

Ammonia in River and Springs. Mean ammonia concentrations were at representative groundwater levels (0.5 mg/l) or below detection levels in samples collected upstream from the

EMF site. Ammonia was detected in Batiste Spring as part of the groundwater monitoring program. Downstream from the EMF site, in the gaining reach of the river, mean ammonia concentrations were highest at SW12 (3.4 mg/l) and decreased further downstream. Elevated concentrations of ammonia at SW12 were attributed to the STP discharge. These observations agree with the STP bioassessment of the Portneuf River (City of Pocatello, 1989). The ammonia introduced into the surface water via Batiste Spring was intermittent, and samples collected along the spring drainage channel at SW11 did not contain detectable levels of ammonia, indicating the total ammonia contribution at Batiste Spring was not high enough to be measurable at points downstream.

Nitrate in Springs. Nitrate was detected at spring sampling stations at mean concentrations ranging from 1.40 to 4.44 mg/l (Table 4.5-4 and Figure 4.5-9). The highest mean nitrate concentrations were found at Batiste Spring (4.44 mg/l), Swanson Road Spring (2.64 mg/l), STP Spring (3.41 mg/l), and Papoose Spring (2.98 mg/l) (Table 4.5-4). Mean nitrate concentrations were lower at sampling points in the drainage channels of Batiste Spring and Papoose Spring (Table 4.5-4).

Note that the STP spring (SW13) has Portneuf River Valley hydrogeochemical characteristics and is located along the east bank of the river. EMF-related groundwater does not impact this spring.

Individual nitrate results for each spring-related sampling point for each sampling round during the RI are shown in Figure 4.5-10. Nitrate concentrations for the springs in the East Side System were generally above 3 mg/l. Slightly elevated nitrate concentrations were detected at the spring within the STP operations area (SW13).

The nitrate concentration of 11 mg/l at Batiste Spring in the April 1993 sample may represent a unique or intermittent event that impacted groundwater and, subsequently, Batiste Spring (Figure 4.5-10). During April 1993, total phosphorus and sulfate at Batiste Spring were also elevated above levels found in previous rounds of sampling (Appendix U).

Nitrate in River. Nitrate concentrations were consistently higher in the gaining reach than in the losing reach of the river (Table 4.5-5). Representative groundwater is a potential source of nitrate in the gaining reach (Figure 4.5-11), due to the relatively high levels of nitrate found in background Michaud Flats and Portneuf River Valley groundwater. The representative nitrate concentrations were 5.52 mg/l and 4.0 mg/l in these two hydrogeochemical regimes.

To the east of the Portneuf River, nitrate in groundwater (3.0 to 3.4 mg/l in Wells 512 and 513) may be related to agricultural activities on the Portneuf River floodplain or to private septic systems. To the west of the river, similar nitrate levels might also be associated with agricultural activities throughout the Michaud Flats, private septic systems, and the land application of sewage sludge in an area north of I-86.

River station SW17 had consistently higher concentrations of nitrate than other losing-reach stations (up to 1.62 mg/l in October 1992). Nitrate levels in the river at SW17 were attributed to the FMC IWW ditch outfall. The maximum nitrate concentration found in a Phase II sample at SW17 was 0.72 mg/l, and the mean concentration in the Phase II samples was 0.57 mg/l. Therefore, it appears that the nitrate concentration detected at SW17 during Phase I resulted from the IWW discharge of nitrate-containing background groundwater (Tables 4.5-1 and 4.5-1a).

As shown in Figure 4.5-11, the highest nitrate levels detected for three out of the four sampling events were detected in samples collected at the downstream stations SW07E to SW01. SW01 was the furthest downstream river sampling station in the RI sampling program. During July 1992, nitrate concentrations for SW01 (2.8 mg/l) and the next station upstream, SW03 (2.7 mg/l), were high compared with other gaining-reach stations.

Water quality sampling conducted by Perry (1977) found that the annual mean concentration of nitrate-N was the greatest at Siphon Road Bridge. This location is the same as RI sampling station SW03 and was the furthest downstream location sampled during Perry's 1975 investigation.

Elevated nitrate concentrations were also detected at stations 5E and 5F (Table 4.5-4), with mean concentrations of 2.47 and 2.56 mg/l, respectively. These stations are located below the Papoose Springs Fish Farm.

In summary, non-EMF activities have increased nitrate concentrations in groundwater that discharges to the Portneuf River, thereby increasing the overall nitrate concentrations in the river. Additionally, nitrates may form as the ammonia discharged from the STP is oxidized, further increasing the nitrate concentrations downstream from the STP. Nitrates are also discharged to the river via groundwater from the EMF site; however, these nitrate loadings are not sufficient to increase the nitrate concentrations along the entire gaining reach of the river (Section 5.4).

Orthophosphate and Total Phosphorus in Springs. Mean orthophosphate and total phosphorus concentrations were at or near the detection limit (0.03 mg/l) at springs SW13, SW09, SW07, SW06, SW04, and SW02 (Figures 4.5-12 through 4.5-15). Orthophosphate concentrations in representative groundwater ranged from 0.06 mg/l to 0.27 mg/l in the three hydrogeochemical regimes. Total phosphorus ranged from 0.15 mg/l to 0.33 mg/l in the three regimes (Table 4.5-4).

Orthophosphate and total phosphorus concentrations were highest at Batiste Spring (SW14), with mean orthophosphate at 2.36 mg/l and mean total phosphorus at 2.71 mg/l. Concentrations decreased downstream along the Batiste Spring drainage channel as evidenced by the mean concentrations of 0.59 and 0.48 mg/l at SW11. Mean orthophosphate and total phosphorus concentrations at Swanson Road Spring (SW15) were 0.99 and 1.05 mg/l, respectively. These levels also exceeded representative groundwater levels.

Orthophosphate and Total Phosphorus in River. Total phosphorus and orthophosphate concentrations were higher in samples collected from the gaining river reach compared with the losing reach (Table 4.5-5 and Figures 4.5-12 through 4.5-15). Although concentrations were generally very low, total phosphorus was present in groundwater beneath the EMF operations areas. Shallow monitoring well 503 near the west bank of the Portneuf River had elevated levels

of total phosphorus that can be attributed to the EMF facilities. Elevated mean total phosphorus concentrations in Swanson Road and Batiste Springs are attributed to the EMF facilities.

Relatively high mean total phosphorus concentrations (0.22 mg/l) were found at the point where Papoose Spring discharges to the Portneuf River (SW05). This sampling point is downstream of the Papoose Springs Fish Farm. As total phosphorus was not elevated in the spring (SW07) and spring drainage (SW06) above the fish farm, the total phosphorus at SW05 is attributed to the fish farm.

Individual total phosphorus results are shown in Figures 4.5-13 and 4.5-15. These figures show that total phosphorus concentrations in the gaining reach of the Portneuf River were consistently highest at SW12. This river sampling station is located at the STP discharge and is upstream from where the Batiste System discharges into the river.

Water quality sampling conducted by Perry (1977) found that Pocatello STP effluent had much higher concentrations of total phosphorus (8.2 mg/l) compared to other effluent sources to the Portneuf River.

Losing-reach sampling station SW17 had mean total phosphorus concentrations (0.64 mg/l) above those detected in samples collected at the gaining-reach river sampling stations (Table 4.5-5). The total phosphorus concentration measured at station SW17 (0.64 mg/l) was likely attributable to discharge of background groundwater and IWW ditch water that may be slightly elevated in total phosphorus. Phase II sampling conducted at SW17 showed mean total phosphorus concentrations of 0.14 mg/l, lower than the Phase I findings (Tables 4.5-1 and 4.5-1a).

Fluoride in Springs. Mean fluoride concentrations for all 12 spring sampling points ranged from 0.3 to 0.8 mg/l (Table 4.5-4). Representative groundwater fluoride concentrations were 0.6 mg/l for Bannock Range groundwater, 0.8 mg/l for Michaud Flats groundwater, and 0.41 mg/l for Portneuf River Valley groundwater. Historical analysis of fluoride in springs

(Perry, et. al., 1990) indicated that the Papoose springs generally had higher fluoride than springs closer to the EMF facilities.

In characterizing the spring groups, Perry et al. (1990) found fluoride concentrations to be significant. Historically (from 1978 to 1980), fluoride concentrations were four to five times greater (1.32 mg/l) in the Papoose System compared with fluoride concentrations in the other three spring groups (0.30 to 0.44 mg/l). During the RI, the highest mean fluoride concentrations (0.7 and 0.8 mg/l) were still found in Papoose System springs. The two East Side System springs (SW13 and SW9) had fluoride concentrations in the 0.30 to 0.44 mg/l range. Mean fluoride concentrations for Batiste Spring (0.6 mg/l) and Swanson Road Spring (0.5 mg/l) were greater than the historical means for these East Side springs and were comparable to fluoride concentrations (0.5 and 0.6 mg/l) in the Papoose Springs (stations SW07, SW06, and SW05). The Papoose Spring System is not impacted by the EMF facilities

Fluoride in River. Both losing- and gaining-reach river sampling stations had fluoride concentrations below 0.5 mg/l. The Phase I sample from SW17 contained 0.7 mg/l of fluoride, similar to the fluoride levels in background groundwater that is discharged via the IWW ditch. Based on Phase I findings, the elevated fluoride level in the river at SW17 may be attributable to the IWW ditch outfall. Subsequent sampling conducted at SW17 showed a decrease in fluoride at this station with a Phase II mean concentration of 0.3 mg/l (Tables 4.5-1a and 4.5-5).

Sulfate in Springs. As indicated in Section 4.4, above-representative level mean concentrations of sulfate in Swanson Road Spring (104 mg/l) and in Batiste Spring (113 mg/l) were attributed to EMF sources (Figure 4.5-16). Sulfate concentrations were also consistently higher in gaining-reach river water (ranging from 54 to 70 mg/l) than in losing-reach river water (38 to 45 mg/l), indicating that the groundwater recharging the river contains higher sulfate concentrations than the upstream river water. However, the overall increase in sulfate concentrations downstream of the EMF facilities was not solely attributable to the EMF-derived sulfate discharges at the springs.

Sulfate in River. Figure 4.5-17 shows that sulfate concentrations in the gaining reach of the Portneuf River were generally highest at SW12 (mean concentration of 65.4 mg/l). This river sampling station is located at the STP discharge and is upstream of the Batiste Spring discharge point. The STP contributes to the higher levels of sulfate in the gaining reach of the Portneuf River.

As seen in Figure 4.5-17, river sampling station SW17 had generally higher sulfate concentrations than other losing-reach river sampling stations. The Phase I sulfate results at SW17 are indicative of the sulfate in groundwater discharged via the IWW ditch to the river. Phase II sampling indicated a mean sulfate concentration of 35 mg/l at SW17. The Phase II sulfate levels were comparable to other losing-reach river sampling stations, which ranged from 38 to 45 mg/l in Phase I.

Radiological Parameters in River and Springs

Surface water analytical results for gross alpha, gross beta, radium-226, radium-228, and uranium-233/234 are discussed in this section. Samples were also tested for uranium-235 and uranium-238, and neither isotope was detected.

To assess the nature and extent of radiological parameters in springs that could be attributed to the EMF facilities operations, sampling results from spring sampling stations were compared with each other. Since EMF-affected groundwater enters the surface water system at Swanson Road and Batiste springs, results for these two springs were compared to the other springs in the study area (Table 4.5-6).

With respect to the Portneuf River, radiological parameters in surface water samples collected from the gaining reach were compared with those collected from the losing reach. Particular consideration was given to sampling stations between SW16 and SW20 (in the vicinity of the EMF facilities), including SW17, located at the FMC outfall (Table 4.5-7). A tabulation of all radiological analyses for each surface water sample collected during the RI is presented in Appendix U.

Gross alpha, radium-226, and radium-228 activities in springs and spring drainages revealed no discernible trends that would indicate potential anthropogenic impacts. The maximum gross alpha activity among all of the springs was detected at SW07 (Papoose Spring) at an activity of 8.84 ± 2.30 pCi/l. Gross alpha activity in Batiste and Swanson Road Springs was comparable to gross alpha activity in other springs.

Radium-226 was detected in three samples from SW07, with activity ranging from 1.40 ± 0.38 pCi/l to 1.93 ± 0.52 pCi/l. A radium-226 activity of 5.20 ± 0.26 pCi/l was detected in SW05 during the October 1992 sampling event. Radium-226 was also detected at SW11 at an activity of 2.60 ± 0.40 pCi/l and at SW15 with activity measurements of 1.50 ± 0.62 pCi/l and 1.82 ± 0.25 pCi/l.

Radium-228 was not detected at SW14, SW11, and SW09. In other spring sampling locations, radium-228 was detected in at least one round. At SW15, radium-228 activity was comparable to the activity detected at other spring sample stations. The highest activities of radium-228 were measured at SW04 (3.5 ± 0.9 pCi/l) and SW02 (5.3 ± 1.2 pCi/l).

Gross beta radiation was detected at every spring during every round of sampling. No single sampling event consistently exhibited the highest gross beta activities. Most if not all gross beta radiation in spring samples are believed to be attributable to potassium-40 (K^{40}), a beta emitter. Using the detected concentration of potassium, the activity of K^{40} was estimated for each sample. The natural radioactive decay calculated from K^{40} -derived beta emissions as a percentage of the gross beta emissions measured in the spring samples is presented in Table 4.5-8.

Samples from selected springs (SW14, SW13, and SW05) were analyzed for uranium isotopes during the February 1993 round of sampling only. Uranium-233/234 was detected in all three samples at similar levels (1.08 ± 0.27 , 1.67 ± 0.52 , 1.19 ± 0.32 pCi/l, respectively). SW14 (Batiste Spring) is known to be impacted by EMF-related constituents and the other two springs

are not impacted. The levels of uranium-233/234 detected in all three springs are considered representative of unimpacted groundwaters. Uranium-235 and uranium-238 were not detected.

Gross alpha radiation was detected at all river sampling station sampling points sampled during the February 1993 round of sampling (Table 4.5-7). Gross alpha radiation was also detected in two or three rounds of sampling at SW25, SW23, SW22, SW20, SW19, SW16, and at all downstream river sampling stations.

Gross beta activities showed moderate variations from station to station, with no discernible trend indicating anthropogenic impacts. Gross beta radiation at river sampling stations was detected at every sampling point during every round of sampling, with the exception of SW21 during April 1993. Over three sampling events, upstream river stations had higher activity than downstream stations. The two highest measurements of gross beta activity (12.00 ± 2.00 and 13.80 ± 4.31 pCi/l) were at SW01 and SW23, respectively. However, these two stations also had the lowest activities of gross beta in other rounds of sampling. In general, gross beta levels appeared to decrease from the furthest upstream river location (SW25) to the furthest downstream locations (SW03 and SW01).

As in spring samples, a large percentage of gross beta radiation in river water samples is attributable to the natural abundance of K^{40} . Table 4.5-8 presents the natural radioactive decay calculated from K^{40} -derived beta activity as a percentage of the gross beta measured in the surface water samples. It is apparent that most, if not all, beta radiation can be attributed to the naturally occurring radioisotope K^{40} in the Portneuf River water.

Radium-226 was detected in one sampling round at stations SW25, SW24, SW19, SW12E, and SW01 and in two sampling rounds at stations SW17 and SW03. Radium-228 was detected at least once in all upstream river stations except SW22 and was detected in three rounds of sampling at SW23, SW21, and SW20. Radium-228 was detected during one round of sampling at the downstream stations SW12E, SW12, SW07, and SW01. Results for both radium-226 and

radium-228 indicate only moderate variations, with no clear distinctions between losing-reach and gaining-reach stations.

Samples from stations SW25, SW24, SW22, SW17, SW10, and SW1 were analyzed for uranium isotopes during the February 1993 round of sampling only. Uranium-233/234 was detected at comparable activities (1.12 ± 0.47 to 1.40 ± 0.35 pCi/l) in all six samples. Uranium-235 and uranium-238 were not detected.

4.5.2 NATURE AND EXTENT OF EMF-RELATED CONSTITUENTS IN SEDIMENTS

The nature and extent of EMF-related constituents in sediments were investigated using statistical comparisons of constituent concentrations in different sample groups, cluster analysis, and direct comparison of sediment chemical concentrations to representative soil concentrations. The results of these comparisons and analyses were used to draw conclusions as to the nature and extent of EMF effects on river and spring sediments. The highest degree of confidence was placed on the statistical analyses. The comparison of sediment concentrations with soil concentrations is a more qualitative comparison because sediment chemistry is not directly comparable to surface soil chemistry.

In two locations, SD11 and SD9, silt and clay-rich sediments were collected in a spring pools with very low current velocities. These two locations are within the area of surface soils that have been influenced by EMF emissions. The EMF effects resulting from surface runoff pathways and aerial deposition pathway, if significant, would likely have been reflected in these sediment samples.

4.5.2.1 Sediment Chemistry Data – Overall Results

The only sediment sample that directly reflected a release from the EMF facilities was SD17, collected at the IWW ditch outfall. The investigation in the area of the outfall demonstrated a very localized effect. Statistically significant elevated chemical concentrations were not encountered at sample locations further downstream.

Cadmium, chromium, vanadium, zinc, fluoride, and total phosphorus were detected in sediment sample SD17 at concentrations in excess of the upstream sediment concentrations and representative soil concentrations (Tables 4.5-9 and 4.5-10; Figures 4.5-18 and 4.5-19). Sediment samples collected downstream from SD17 in the river channel and spring drainages did not contain elevated concentrations of cadmium, chromium, vanadium, zinc, or fluoride, and one downstream sample, SD10, had higher total phosphorus than SD17 (Tables 4.5-11 and 4.5-12; Figures 4.5-20 and 4.5-21).

Sample SD10 contained 7,150 mg/kg total phosphorus, the highest of any sediment sample. This sample was collected from the river channel where it is joined by the Batiste Spring drainage. The next highest concentration of total phosphorus was found at the IWW ditch outfall (5,340 mg/kg). Concentrations above the upstream sediment and representative soil levels were observed in the Fort Hall Bottoms (SDC1 at 1,160 mg/kg and SDC4 at 1,060 mg/kg).

Arsenic values exceeded upstream sediment and soil representative levels at stations SD18 (8.4 mg/kg) and SD8 (9.9 mg/kg). Spring sediment samples that exceeded the upstream sediment and representative soil level for arsenic were samples SD4 at Siphon Road Spring (8.2 mg/kg), SD7 at Papoose Spring (9.1 mg/kg) and SD2 at Twenty Spring-East (13.8 mg/kg). The Papoose Spring sample was taken in a ponded water area with very low energy, and the sample from Twenty Springs was taken in a low-energy swampy area. Like the river sediments, neither of these stations contained the suite of metals, fluoride, and total phosphorus associated with the EMF facilities. Therefore, the constituents found in these sediments are not reflective of EMF effects.

The highest levels of lead were detected in upstream sampling locations SD23 and SD24 (71.9 mg/kg and 51.6 mg/kg) respectively (Figure 4.5-22). Given the upstream locations of these samples relative to the EMF facilities, it is clear the lead is not related to the EMF facilities. The next highest lead concentrations in river sediment were found at locations SD19 and SD20. In general, lead concentrations were higher in upstream samples than in downstream samples.

Figure 4.5-23 displays the lead values detected in the spring sediments. The discharge point of the Papoose System contained the highest level of lead detected in spring sediment (50.5 mg/kg).

Mercury was detected in one upstream location, SDA1, at a concentration of 0.55 mg/kg and one downstream location, SDB1, at a concentration of 1.1 mg/kg (Figure 4.5-22). The occurrence of mercury in the river sediments does not appear to be related to any identified specific source along the river and may, in fact, be naturally occurring (Appendix Q.)

Gross alpha activities appear to be related to soil textures, with sediments rich in clay or gravel being generally higher than those containing silt or sand. Sample location or proximity to the EMF site does not appear to be a factor. The one exception was at the IWW ditch outfall, which had the highest level of gross alpha activity (29.2 ± 3.6 pCi/g).

Gross beta activities were positively correlated with potassium-40 content, with some exceptions. Gross beta activities were less than representative soil levels.

4.5.2.2 Sediment Statistical Comparisons

Sediment samples were compared statistically using several different methods: t-tests, non-parametric ANOVA (analysis of variance), and cluster analysis. The student's t-test uses the reported concentrations of chemicals and allows for a one variable (chemical) comparison between two groups of samples. It assumes a normal distribution. The objective in performing a student's t-test was to investigate differences between results for statistical significance. The non-parametric ANOVA is a test that is independent of the population distribution and the presence of nondetects in the dataset. The non-parametric ANOVA highlights differences that may be present, although masked by nondetects or other "noise" in the dataset. The cluster analysis compares sediment samples using numerous analytes concurrently, whereas the t-tests and ANOVA can only be applied to one analyte at a time.

Student's t-test. Sediment samples were assigned to spring, upstream, near-site, and downstream groups for statistical comparisons (Table 4.5-13). The spring and near-site samples

were placed into separate groups to ensure that any influences from the EMF site would be identified in the statistical tests. Samples in the spring group are SD2, SD4, SD5, SD6, SD7, SD9, SD11, SD13, SD14, and SD15. Samples in the near-site group most likely to reflect the cumulative effects of IWW ditch outfall, surface runoff, and direct aerial deposition to the river are SD16, SD18, SD19, and SD20. Samples SD21, SD22, SD23, SD24, SD25, SDA1, and SDA2 form the upstream group, which is least likely to be affected by EMF-related activities. The downstream sediment sample group includes SD12, SD10, SD8, SD3, SD1, SDB1, SDC1, SDC2, and SDC4. Sample SD17 was not included in any group because it reflected EMF-related influences.

The test hypothesis was that the sediment sample groups were collected from the same sediment population. The hypothesis was tested at the 95% confidence level. Where the absolute value of the calculated t-value was greater than the corresponding 95% confidence interval t-value from the statistical table, the hypothesis would be rejected. Rejection of the hypothesis would indicate that the two sample groups were not collected from the same population, and that there is a statistically significant difference between their mean concentrations.

Results of the t-tests show that sample means for near-site sediments are not statistically different from upstream sediment means for any constituents, except iron (Table 4.5-13). Iron is the only constituent for which there was a statistically significant difference between the sample means of the two sediment groups, with upstream sediments having a higher iron content.

When spring sediments were compared with the upstream sediments, the upstream sediments had higher mean concentrations of aluminum, copper, lithium, manganese, and nickel. Spring sediments were higher in beryllium, which is likely a result of the elevated beryllium concentrations in samples SD9 (FMC Employee Park) and SD2 (Twenty Springs East).

Upstream sediments had statistically higher concentrations of cobalt, manganese and vanadium at the 95% confidence level, compared with spring sediments. This result is particularly

important for vanadium given that it is a characteristic constituent of EMF potential source materials (e.g., ore and precipitator dust).

There was no statistically significant difference in total phosphorus content between upstream and downstream sediment even though the mean total phosphorus concentration in the downstream sediment sample group was 1,463 mg/kg compared with a mean in the upstream sediments of 357 mg/kg. This indicates that, although there was a higher concentration of total phosphorus at SD10, the overall total phosphorus content of downstream sediments is not statistically different from the upstream sediments.

Non-Parametric ANOVA. A non-parametric test, instead of a t-test, was used to evaluate selenium, mercury, thallium, and cadmium because these datasets contained a high proportion of nondetects. The nonparametric ANOVA or Kruskal-Wallis tests whether any of the sediment sample groups are from a different population. This is tested at the 95% confidence level. The same sediment groups used in the t-test analyses were used in the ANOVA analysis. The test results indicate no differences between sediment sample groups for mercury, selenium, thallium, or cadmium (Table 4.5-13).

Molybdenum was detected in only one upstream sample, and antimony was not detected in any sediment sample. Therefore, neither of these parameters were tested for significance using either the t-test or non-parametric ANOVA.

Cluster Analysis. A cluster analysis was performed on the sediment data for manganese, aluminum, iron, total phosphorus, fluoride, zinc, barium, arsenic, gross beta, lead, cadmium, chromium, copper, vanadium, and selenium. These analytes were selected because these analytes best encompassed the overall dissimilarities in the sediment composition. Note that the six characteristic constituents are included in this grouping. Other metal and radiological constituents (e.g., Ni, Li) correlated well with one or more of the constituents used in the cluster analysis, and would only have served to reduce the “dissimilarity” between samples had they been included.

The results indicate that SD17 (Phase I IWW outfall) and SD17A (Phase II IWW outfall) are very dissimilar to other samples, and are not very similar to one another. SDB1A, collected from a public boat launch area, is also very distinct from other samples. According to the cluster analysis, it is most similar to SDA1, which was collected several thousand feet upstream from the EMF site. If SDA1 does not reflect any EMF-related impacts (which is likely since there is no pathway between the site and SDA1), then it follows that the chemistry of sample SDB1A does not necessarily reflect EMF-related influences. Why these two samples are different from the others cannot be explained with the available data.

Sample SD10, which contained the highest total phosphorus concentration, is most similar to SD13, the sediment sample collected at the STP spring pond. If SD10 were indicative of EMF-related impacts to the river, SD10 might have been expected to be more similar to SD14, SD17, SD17A, or SD15, because these samples were collected in the immediate vicinity of EMF-related discharges.

Upstream sediments are not similar to each other and show the same degree of similarity to downstream and spring sediments. Because upstream sediments, downstream sediments, and spring sediments do not show distinct groups that are spatially related to EMF discharges or transport pathways, the cluster analysis demonstrated that there is no distinct EMF fingerprint in the sediments.

4.5.2.3 River Sediments – Detailed Discussion

A detailed discussion of the chemical characteristics of river sediments upstream of all EMF-related discharges to the Portneuf River is presented below. This characterization of upstream sediments provides a basis for evaluating the analytical results for sediment samples collected in areas that might have been influenced by pathways that transport EMF-derived constituents to surface water sediments. This characterization of upstream sediments is followed by a sample-by-sample discussion of sediment samples collected in the Portneuf River channel. Conclusions regarding EMF-related influences are based on the results of the statistical tests

(Section 4.5.2.2), comparisons with upstream sediment concentrations and soil representative levels, the presence of characteristic EMF parameters, and the presence of a pathway between the EMF facilities and the river sediment and depositional environment. A sample-by-sample discussion of the spring sediment samples is presented in Section 4.5.2.4.

Upstream Sediments (SD25 to SD21, SDA1 and SDA2). Upstream sediment samples exceeded the soil representative levels for aluminum (SDA1), boron (SDA1), copper (SDA1 and SD23), lead (SD24 and SD23), manganese (SD25), mercury (SDA1), molybdenum (SD21), and zinc (SDA1) (Table 4.5-9). The upstream sediments did not contain orthophosphate or total phosphorus at concentrations in excess of the representative levels. Fluoride exceeded its representative level of 600 mg/kg in SD23 (1,300 mg/kg). Despite these differences, it appears that the upstream sediments were generally similar in chemical composition to local soils. Slightly higher zinc, copper, mercury, and lead concentrations may be due to discharges to the river from potential sources within Pocatello or further upstream. Alternatively, the higher concentrations may be indicative of natural variability within the river system. Regardless, the upstream sediment metal concentrations were similar to soil representative levels.

Sample SD20. Location SD20 is approximately 1,800 feet downstream from SD21 (Figure 4.5-1f). The texture of sample SD20 was a sand with silt and gravel. Lead (61.0 mg/kg) and silver (3.0 mg/kg) exceeded representative levels. All other constituents were within soil representative levels, including the EMF characteristic metals. This location was not impacted by EMF operations.

A comparison of the results found at this location with the upstream samples indicates similar, but generally lower concentrations of metals and nutrients (Tables 4.5-9 and 4.5-10). This result is expected since the sample was a sand rather than a clay, and less likely to contain naturally occurring trace metals in its matrix or to contain adsorbed metals.

Sample SD19. Location SD19 is approximately 1,000 feet downstream from SD20 (Figure 4.5-1f). The texture of this sample was a silty clay. Lead (38.6 mg/kg) and copper

(12.7 mg/kg) concentrations exceeded soil representative levels. Higher lead, copper, and fluoride concentrations were detected in other samples further upstream, so their occurrence here does not indicate an EMF-related impact. This sample does not appear to be indicative of EMF facility impact.

Sample SD18. Location SD18 is near the old FMC and Simplot outfalls. It is approximately 350 feet downstream from SD19. The texture of the sediment was sand with gravel. Arsenic was detected at 8.4 mg/kg, above the representative soil level of 7.7 mg/kg. Thallium was detected at a concentration of 0.30 mg/kg compared with a representative level of 0.27 mg/kg. The remaining constituents were below both representative and upstream trace metal levels (Table 4.5-9). The arsenic concentration is likely within the variability of representative levels in the river sediments. Because the reported thallium value was an estimated value (i.e., J qualifier) that is very close to the representative level for soils, thallium was not considered elevated.

Sample SD17. Sample location SD17 is located several feet beyond and downstream of the current FMC outfall. Its texture was a sandy clay. When dried, the material contained a gray, clay-like material with shell and rock fragments. The sand fraction was coarse, pink and purple sand. Also, the sample contained considerable organic matter in addition to the mineral matrix. This sample contained a number of constituents above representative soil levels, including the suite of constituents characteristic of potential sources at FMC (Tables 4.5-9 and 4.5-10). Therefore, the sample is considered to have been influenced by FMC industrial activities. A petrographic thin section of the sample was made and compared with thin sections of slag, phosphate ore, and precipitator slurry. The visual microscopic comparison indicated that the sediment sample contained components of precipitator dust and ore. The presence of ore is not surprising since the IWW ditch runs just to the east of the FMC ore pile. The thin section evaluation report is presented in Appendix I.

During Phase II sampling in July 1993, three additional samples were gathered in the vicinity of the FMC outfall (Figure 4.5-1h). Sample SD17A was collected from the river channel directly in

front of the FMC outfall pipe. FMC had placed a steel plate in front of the pipe to act as a baffle, and the sample was taken behind it. Sampling in front of the pipe was not possible because there was very little space between the plate and the outfall pipe, and the river bottom area had been thoroughly scoured. The texture of the sample taken from this area was a sandy gravel. It contained above-representative levels of various parameters including the suite of FMC characteristic constituents (Table 4.5-14). Sample SD17B was taken downstream on the eastern side of the river (the main channel is along the west side of the river). Its texture was a fine sand with some shell fragments. Sample SD17C was taken on the east border of the river approximately 70 feet (22 m) downstream from the outfall pipe. Its texture was moderate to fine sand with shell fragments. All parameters for samples SD17B and SD17C, with the exception of calcium (102,000 mg/kg and 208,000 mg/kg, respectively), were below representative soil levels, and contained no evidence of the EMF characteristic constituents. The high calcium level were probably due to the dissolution of the shells during sample preparation.

The sampling carried out in the area of the FMC outfall indicates a very localized impact on river sediments around the outfall. Samples collected in the downstream portion of this area and at points further downstream did not contain the EMF characteristic constituents above representative levels and, hence, indicate that there has been no measurable impact beyond the outfall.

Sample SD16. Location SD16 is located north of Batiste Road. The sediment sample was taken on the eastern side of the river. Its texture was silty clay. Copper (30.8 mg/kg), thallium (0.73 mg/kg), and zinc (56.9 mg/kg) were detected in the sample at above representative soil values (Table 4.5-9). The remaining parameters were below representative soil concentrations and upstream sediment concentrations. The absence of high cadmium, chromium, and vanadium, and the low values of fluoride (273 mg/kg) and total phosphorus (554 mg/kg) indicate that the sediments were not impacted by the EMF facilities.

Sample SD12. Location SD12 is located 80 feet (24 m) downstream from the STP discharge. Sediments were collected on the west side of the channel. The sediment texture was

sand. Beryllium (1.1 mg/kg) and silver (2.2 mg/kg) exceeded representative soil concentrations. All other parameters were within both the representative soil range and upstream sediment sample values (Tables 4.5-9 and 4.5-10). It should be noted that, given the geometry of the river at this location, SD12 is probably not an area where deposition from the STP discharge would occur.

Sample SD10. Sample SD10 was collected within the river just downstream from the mouth of Batiste Spring. The texture of the sample taken here was fine sand. Metals, orthophosphate, and fluoride concentrations were below the representative soil concentrations and upstream sediment concentrations (Tables 4.5-9 and 4.5-10). The only constituent above its representative level was total phosphorus, which had a value of 7,150 mg/kg. As illustrated by the statistical test, total phosphorus concentrations in downstream sediments were not significantly higher. The cluster analysis indicated that sediments at SD10 were most similar to those at the STP Spring (SD13).

Sample SD8. Location SD8 is near the mouth of the spring-fed pond at the FMC park (Figure 4.5-1d). The texture of the sample contained considerably more silt and clay than sand. Arsenic was detected in this sample at a concentration of 9.9 mg/kg, compared to a representative soil concentration of 7.7 mg/kg (Table 4.5-9). Although this arsenic concentration may reflect an anthropogenic impact to the Portneuf River, other EMF-related constituents did not exceed representative concentrations.

Sample SD3. Sediment sampled at location SD3 was taken in the river at the bridge at Siphon Road (Figure 4.5-1b). Its texture was loam with sand and gravel. The relatively low aluminum value (3,670 mg/kg) suggests that the portion of the sample tested in the laboratory was more sandy than silt/clay. None of the analytical parameters exceeded representative soil concentrations; all concentrations were below the values found in the upstream samples (Tables 4.5-9 and 4.5-10).

Sample SD1. Sediment sample SD1 had a texture described as loam (e.g., approximately equal portions of sand, silt, and clay). The only parameters that exceeded representative soil concentrations were silver and thallium. Silver was detected at a concentration of 2.1 mg/kg compared with a representative level of 1.9 mg/kg, and thallium was detected a concentration of 0.28 mg/kg compared with a representative level of 0.27 mg/kg. The remaining parameters were within representative levels and generally below those values found in the upstream samples.

Sample SDB1. Location SDB1 sediment was collected at a public boat launching area in the Fort Hall Bottoms and above the high water mark of the American Falls Reservoir (Figure 4.5-1i). Its texture was silty clay. This sample contained numerous parameters above representative soil levels although not generally above values found in the samples upstream of the EMF facilities (Tables 4.5-9 and 4.5-10). Also, not all of the EMF characteristic constituents were present. The lack of elevated levels of vanadium, cadmium, total phosphorus, and fluoride indicates the absence of EMF facilities-related particulates (Tables 4.5-9 and 4.5-10). The presence of trace metals, such as lead (30.9 mg/kg), copper (25.5 mg/kg), mercury (1.1 mg/kg), and zinc (97.1 mg/kg), at above-representative levels may be attributed to high clay content. Other constituents above representative levels were aluminum (16,200 mg/kg), iron (16,100 mg/kg), and total organic carbon (11,074 mg/kg). Aluminum and iron concentrations reflect the high clay content of this sediment. This content, combined with the high organic content, imply a potential for a high metal adsorption/absorption capacity of the soil matrix. Furthermore, the Fort Hall Gravels which outcrop in this area contain native elemental and mineral-phase mercury.

Sample SDC1. Sediment sample SDC1 was taken on the downstream side of a point bar. Its texture was silt with fine sands (Figure 4.5-1i). With the exception of calcium (166,000 mg/kg) and total phosphorus (1,160 mg/kg), all parameters were below representative levels and, in general, below upstream sample levels (Tables 4.5-9 and 4.5-10).

Sample SDC2. River sediment was sampled at location SDC2, approximately 1,000 yards downstream from SDC1 (Figure 4.5-1i). Its texture was silty clay. With the exception of

calcium (88,500 mg/kg), all parameters were below representative soil levels (Tables 4.5-9 and 4.5-10).

Sample SDC4. Sediment sample SDC4 was taken approximately 400 feet downstream from SDC2 (Figure 4.5-1i). Its texture was a clayey silt. Boron and copper were slightly above representative soil levels. Boron was reported at 13.1 mg/kg compared with a representative level of 12.8 mg/kg, and copper was detected at 12.9 mg/kg compared with a representative level of 12.6 mg/kg. Calcium (93,200 mg/kg) and total phosphorus (1,060 mg/kg) were also detected above representative soil levels. Total organic carbon was detected at 9,468 mg/kg.

Radiological Parameters

Gross alpha and gross beta were measured on all sediment samples taken during the investigation. All measurements were below their corresponding soil representative levels.

Gross alpha values ranged from 6.33 ± 2.96 pCi/g (SDA2) to 13.6 ± 1.28 pCi/g (SD23) in the upstream samples (Tables 4.5-15 and 4.5-16). The highest gross alpha activity (29.2 ± 3.6 pCi/g) was found at location SD17, the FMC outfall. This observation is expected since the FMC potential sources (Section 4.2.3) contain alpha emitters. The elevated gross alpha at SD17 corroborates previously discussed evidence of EMF impact at this location.

With the exception of SD17, sediment samples from SD21 to SDC4 all contained 12 pCi/g or less gross alpha, which is less than the high end of the range of the activities detected in upstream samples. Even SDB1, which contained several metals at elevated concentrations, but not those characteristic of EMF potential sources, has a relatively low activity (8.15 ± 3.33 pCi/g). This observation lends additional support to the conclusion that above-representative inorganic parameters found in SDB1 were not related to the EMF facilities.

In summary, the EMF-related discharge responsible for the gross alpha values observed at SD17 does not appear to have impacted sediments further downstream. In addition, the lack of elevated gross alpha activities in river sediments at locations other than SD17 suggest that

impacted offsite surface soils have not migrated to the river as surface runoff. The gross alpha results support conclusions drawn from results for the EMF characteristic constituents.

In examining gross beta values, it should be noted that potassium-40 may be a major contributor to these values. Potassium is generally a major component of natural clay soils and, as has been described previously, it is a major component of several EMF potential sources. However, the gross beta and potassium-40 sediment values (Tables 4.5-15 and 4.5-16) were not always well correlated, indicating another unidentified beta-emitting source. However, all gross beta values, including SD17, were below the representative soil value.

Upstream gross beta activities ranged from 10.2 ± 2.62 pCi/g (SDA1) to 25.3 ± 1.45 pCi/g (SD24). These values reflect the silty/clayey nature of the sediments. The highest activity detected among all the samples was at SD17, where 30 ± 3.15 pCi/g gross beta was detected. This observation is not unexpected since this sediment sample contains EMF-related particulates. Sediment samples collected downstream of SD22, excepting SD17, had gross beta values ranging from nondetect at 5 pCi/g to 16.9 ± 2.35 at SD19. These values support the conclusion that EMF-related impacts are confined to location SD17.

4.5.2.4 Spring Sediments – Detailed Discussion

A sample-by-sample presentation of the spring sediment sampling results is provided in this section. Constituents that exceeded representative soil concentrations are highlighted and discussed. The spring sediments are also compared with the sediments collected from the upstream reach of the Portneuf River. Conclusions regarding EMF-related influences are based on the results of the statistical tests (Section 4.5.2.2), comparisons with representative levels, the presence of characteristic parameters, and the presence of a pathway between the EMF site and the spring.

Sample SD15. Location SD15 is at Swanson Road Spring (Figure 4.5-1e). The sediment texture was a sand with silt. The silver concentration (2.1 mg/kg) exceeded the representative

soil level. Other metals were below representative levels. The total phosphorus concentration was 955 mg/kg, above the representative value of 672 mg/kg. Orthophosphate (4.9 mg/kg) and fluoride (333 mg/kg) were below the representative soil levels.

Samples SD11 and SD14. Sediment samples SD14 and SD11 were collected from Batiste Spring and the spring drainage channel (Figure 4.5-1e). The texture of sample SD14 was sand and gravel. The texture of sample SD11 was clayey sandy gravel sample. Sample SD14 contained above-representative concentrations of copper (13.0 mg/kg), lead (29.5 mg/kg), and barium (324 mg/kg). Sample SD11 contained only one constituent, zinc, at an above-representative value (107 mg/kg). As discussed above, the upstream sediments in the river also had lead and copper concentrations in excess of representative soil levels. This appears to be true of the spring sediments as well. The barium content in Batiste Spring sediments may reflect a localized site-related impact.

The sample from SD11 was not collected in the main Batiste channel, but rather in a low-energy pool that is within the area where aerial deposition of EMF-related materials might be expected to have occurred (offsite soil samples SS45-1C and SS023-1C, Table 4.3-3). As discussed in Section 4.3, surface soil samples (north of the EMF facilities) contained the suite of EMF characteristic constituents. However, sediment sample SD11 did not. Since the characteristic constituents were not evident in the sediment, neither air deposition nor overland runoff appear to have had measurable impacts on sediment, even in an area of quiescent surface water. The quiescence of this surface water body is substantiated by the occurrence of clay in the sediments. Deposition of clays on freshwater substrates requires extremely low current velocities in the overlying water column. If significant quantities of EMF materials were transported via the air pathway to surface water and sediments, the particulates would likely be clay size or smaller (less than 1/256 mm), and extremely low current velocity would be necessary for these particulates to collect in sediments.

Sample SD13. Sample SD13 was collected between the Portneuf River and the Pocatello STP sludge-drying beds (Figure 4.5-1e). The spring, located on STP property, has a fairly large

spring pond with sandy sediment as the substrate. The texture of sample SD13 was sand. The sample (Tables 4.5-11 and 4.5-12) contained above-representative levels of total phosphorus (3,950 mg/kg), fluoride (800 mg/kg), and selenium (3.5 mg/kg). The most probable source for the elevated constituents is the STP sludge drying beds. As described in Section 4.4, the springs along the eastern side of the river do not discharge any groundwater impacted by EMF-related activities, thus eliminating the possibility that selenium, total phosphorus, or fluoride in the sediment sample is from the EMF site.

Sample SD9. Sediment sample SD9 was taken at the spring-fed pond at the FMC park (Figure 4.5-1d). Its texture was a loam. This spring is fed by the Portneuf River Valley hydrogeochemical regime and is uninfluenced by EMF facilities-related groundwater because the spring is located on east side of the river. Beryllium (1.40 mg/kg) was above the representative soil level of 1.0 mg/kg. As was true with SD11, this very quiet pond is also within the influence of potential air deposition from the EMF facilities, as shown by impacted offsite soil sample 000-2A (Table 4.3-3), and as was true at SD11, there was no measurable evidence of an EMF-related impact in the sediment. This observation further supports the conclusion that neither air deposition nor overland runoff is a pathway for sediment impact.

Samples SD5 and SD7. Sampling stations SD7 and SD5 are located in the Papoose Spring System (Figure 4.5-1c). Neither spring is downgradient of sources impacting groundwater within the EMF facilities. Sediment sampled at location SD7 was taken in the northeastern portion of the pond fed by Papoose Spring. Its texture was clayey, sandy gravel. Sediment sample SD5 was taken at the mouth of the spring reach as it entered the Portneuf River. Its texture was silty clay. There was an operating fish farm between the two sampling points at the time of sampling. The only parameter with an above-representative concentration in SD7 was arsenic at 9.1 mg/kg. The arsenic representative level for soils is 7.7 mg/kg. The sample collected at station SD5 contained above-representative soil levels of lead (50.5 mg/kg), thallium (0.30 mg/kg), and zinc (54.3 mg/kg). (Soil representative levels are 29.1 mg/kg for lead, 52.8 mg/kg for zinc, and 0.27 mg/kg for thallium.) As noted before, the lead in sediments throughout

the Portneuf River exceeded representative levels at upstream locations, indicating lead is enriched by non-EMF sources in the river sediments relative to soils.

Sample SD4. Sample SD4 was taken at a spring near Siphon Road (Figure 4.5-1b). Its texture was loam. Sample SD4 contained 8.2 mg/kg arsenic, which is greater than the soil representative level of 7.7 mg/kg. The remaining parameters detected in this sample were below representative soil concentrations. There is no groundwater pathway for arsenic transport from the EMF site to the sediments in this spring, making the EMF site an unlikely source of the arsenic detected in this sample.

Sample SD2. Sample SD2 was taken on the eastern branch of Twenty Spring (Figure 4.5-1a). The sediment was silty clay. The area in which it was taken was very swampy. In addition, the recovery for the sample was poor insofar as it was reported to consist of only 20 percent solids. Samples with low percent solids content are difficult to quantitate on a dry weight basis, and the results from such quantitation are generally biased high. Hence, while elements reported as detected in the sample were probably present, their reported values were likely overestimates of the true concentrations. Four constituents were reported at concentrations above representative soil levels. These constituents were arsenic (13.8 mg/kg), beryllium (2.2 mg/kg), chromium (54 mg/kg), and vanadium (192 mg/kg). Zinc, generally found in much greater abundance than vanadium in EMF-related materials, was below the representative soil level at 37.4 mg/kg. Fluoride was detected at 75.3 mg/kg, and total phosphorus was detected at 64.5 mg/kg. These two constituents are considered primary indicators of EMF-related impacts; however, the concentrations of these two constituents in sample SD2 are very low compared to other sediment samples. While the four parameters that exceeded representative levels can be found in EMF potential source-related matrices, the levels of other parameters that have a stronger association with EMF materials suggest that this sample had not been affected by EMF-related activities.

Radiological Parameters

Spring sediments, in general, had higher levels of gross alpha than the river sediments (Table 4.5-16). However, the gross alpha activities in all spring sediments were less than the representative soil level (24.7 pCi/g). There was no correlation of gross alpha activity with location. The highest value (19.8 ± 2.49 pCi/g) was detected at SD14 (Batiste Spring), which is fed by Bannock Range water that is impacted by EMF activities. However, a similar value (14.8 ± 1.35 pCi/g) was detected at SD13, located on the east side of the river and fed by the Portneuf River Valley hydrogeochemical regime, uninfluenced by the EMF facilities. The sediments found in the springs and spring drainage channels are locally derived and have somewhat higher gross alpha activity than the upstream sediment sources.

The same pattern exists for gross beta as exists for gross alpha. The highest gross beta activities were found in the more clay-rich sediments (SD2, SD5, and SD9 at 19.7 ± 2.1 , 18.2 ± 2.3 , and 19.5 ± 2.1 pCi/g, respectively). The gross beta representative soil level is 31.4 pCi/g.

Surface and Subsurface Characterizations
Tables for Section 4.5

Section 4 Nature and Extent of Constituents of Potential Concern

TABLE 4.5-1
COMPARISON OF WATER QUALITY AT RIVER STATION SW17 WITH WELL AND IWW DITCH WATER

Parameter	Production Well Water (unimpacted) FMC-1 ^(a)	IWW Ditch Water (Phase I) FSWTWW01 ^(a)	Surface Water (Phase I) SW17 ^(a)	Surface Water (Phase II) SW17 ^(a)
	10/90 ^(b)	9/92 ^(b)	Mean 7/92 - 4/9/93 ^(b)	Mean 7/93 ^(b)
Alkalinity, bicarbonate (mg/l)	187	324	188.3	196
Calcium (mg/l)	87.6	141J	62.2	64.7
Chloride (mg/l)	96	1620	54.6	—
Magnesium (mg/l)	27.7	22	22.6J	23.8
Potassium (mg/l)	11.2	7470J	8.4	6.9
Sodium (mg/l)	50.1	1450J	47.7	29.7
Sulfate (mg/l)	142	8400	65.4	35
Specific conductance at 25°C, µmhos/cm	925	NA	738	—
pH	7.5	NA	8.52	—
Temperature, °C	13.5	NA	17.6	—
Total dissolved solids (mg/l)	585	7460	403.3J	340
Ammonia (NH ₃ as N) (mg/l)	NA	288	ND	0.5
Nitrate (NO ₃ as N), (mg/l)	1.9	18.4	1.29J	0.57
Orthophosphate (PO ₄ as P), (mg/l)	0.45	2210	0.32J	0.08
Phosphorus, total (mg/l)	0.5	2590	0.64	0.14
Fluoride (mg/l)	0.4	61.6	0.65	0.3
Aluminum (mg/l)	NA	7.66J	1.02	1.88
Antimony (mg/l)	NA	ND	ND	ND
Arsenic (mg/l)	0.0115	1.759J	0.0063J	0.0029
Barium (mg/l)	0.113	0.27	0.081	0.114
Beryllium (mg/l)	NA	ND	ND	0.001
Boron (mg/l)	NA	4.53	0.38J	0.09
Cadmium (mg/l)	0.003	0.0341J	0.0003J	ND
Chromium (mg/l)	0.006	ND	0.002J	0.001
Cobalt (mg/l)	NA	ND	0.008	0.004
Copper (mg/l)	0.004	0.163	0.015	0.007
Iron (mg/l)	0.041	4.99J	0.413	1.465
Lead (mg/l)	0.001	ND	0.002	0.001
Lithium (mg/l)	NA	1.599	0.049	0.027
Manganese (mg/l)	0.0394	0.187	0.023	0.048
Mercury (mg/l)	NA	ND	ND	ND
Molybdenum (mg/l)	NA	0.19	ND	ND
Nickel (mg/l)	NA	0.17	0.013	0.01
Selenium (mg/l)	0.0025	4.217J	0.0053	ND
Silver (mg/l)	0.004	0.035	0.0029J	ND
Thallium (mg/l)	NA	0.318J	ND	ND
Vanadium (mg/l)	0.0061	0.83	0.027J	0.002
Zinc (mg/l)	0.0106	5.25	0.039J	0.020
Gross alpha (pCi/l)	NA	186.50	5.80	1.0
Gross beta (pCi/l)	NA	103.58	4.89	9.90
Radium-226 (pCi/l)	NA	NA	0.94	0.50
Radium-228 (pCi/l)	NA	NA	0.50	0.50

Notes: (a) Sample ID
(b) Sample date

ND = Not detected/below detection limit
NA = Not analyzed

J = Estimated value

Table 4.5-1A
Analysis Results for Water Samples Collected from IWW Ditch

Table 4.5-1A

Parameter	NEIC Sample		Phase II Samples																Mean of Phase II Samples	Well 125	
	NEIC 041W		O307IWA		O307IWD		O307IWE		O307IWF		O307IWG		O307IWH		O307IWN		O307IWP			Range of Values (4/92 - 12/93)	
	Value Reported	Detection Limit	Value Reported	Data Qualifier	Value Reported	Data Qualifier	Value Reported	Data Qualifier	Value Reported	Data Qualifier	Value Reported	Data Qualifier	Value Reported	Data Qualifier	Value Reported	Data Qualifier	Value Reported	Data Qualifier			Minimum Value
Alkalinity, bicarbonate			168		170		170		178		194		188		180		164		176.5	176	199
Alkalinity, carbonate -			0		0		0		0		0		0		0		0		0	0	5
Aluminum, dissolved			0.023	U	0.023	U	0.023	U	0.023	U	0.023	U	0.023	U	0.023	U	0.023	U	0.0115		
Aluminum, total			0.023	U			0.023	U	0.023	U	0.023	U	0.023	U	0.023	U	0.023	U	0.0115	0.023	0.054
Ammonia (NH3 as N)			0.5	U			0.5	U	0.5	U	0.5	U	0.5	U	0.5	U	0.5	U	0.25	0.2	0.5
Antimony, dissolved			0.045	U	0.045	U	0.045	U	0.045	U	0.045	U	0.045	U	0.045	U	0.045	U	0.0225		
Antimony, total	ND	0.06	0.045	U			0.045	U	0.045	U	0.045	U	0.045	U	0.045	U	0.045	U	0.0225	0.039	0.15
Arsenic, dissolved			0.00867		0.00729		0.00979		0.00922		0.01064		0.00638	U	0.00875		0.00847		0.00546	0.0049	0.0066
Arsenic, total	0.017	0.01	0.00686				0.00679		0.00671		0.00739		0.00741		0.00657		0.00633		0.00687	0.00362	0.0096
Barium, dissolved			0.07917		0.08288		0.08637		0.08408		0.08146		0.08375		0.07961		0.08037		0.08221	0.0825	0.0929
Barium, total			0.08255				0.08615		0.08288		0.08397		0.08463		0.08299		0.0855		0.0841	0.076	0.16
Beryllium, dissolved			0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.0005		
Beryllium, total	ND	0.005	0.001	U			0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.0005	0.001	0.003
Boron, dissolved			0.17381	J	0.18236	J	0.18299	J	0.1909	J	0.17254	J	0.17727	J	0.18435	J	0.17555	J	0.17997		
Boron, total			0.16219	J			0.17882	J	0.19426	J	0.18145	J	0.22324	J	0.18744	J	0.23697	J	0.19491	0.054	0.27139
Cadmium, dissolved			0.001	U	0.001	U	0.0033	J	0.0035	J	0.001	U	0.001	U	0.0026	J	0.001	U	0.0013	0.004	0.004
Cadmium, total	ND	0.005	0.001	U			0.00307	J	0.0041	J	0.001	U	0.001	U	0.0028	J	0.001	U	0.0015	0.0007	0.0142
Calcium			63.8283				65.27745		64.20603		64.86902		65.96515		64.52338		63.11859		64.5411	41.6	49.7
Calcium, dissolved			62.90637		64.28277		64.98514		65.02318		64.64879		65.34284		62.9878		62.80162		64.1223		
Chloride			78				88		78		83.97		82.97		80		80		81.5629	28.9	46.1
Chromium, dissolved			0.00214		0.002		0.00223		0.00209		0.00202		0.00209		0.00221		0.00208		0.00211	0.005	0.013
Chromium, total	ND	0.01	0.0028	J			0.00277	J	0.00309	J	0.00282	J	0.00276	J	0.00307	J	0.00282	J	0.00288	0.0018	0.01
Cobalt, dissolved			0.00577		0.00759		0.00847		0.00957		0.0073		0.00949		0.00621		0.00628		0.00759		
Cobalt, total			0.00832				0.00876		0.00847		0.00942		0.00672		0.00876		0.00869		0.00845	0.003	0.027
Copper, dissolved			0.0045	U	0.00484	U	0.01033	U	0.01289	U	0.01033	U	0.01307	U	0.004	U	0.00512	U	0.00407	0.003	0.0074
Copper, total	ND	0.025	0.01111	U			0.01064	U	0.01498	U	0.0158	U	0.01001	U	0.0103	U	0.01271	U	0.00611	0.002	0.0043
Dissolved oxygen			6.2		6.4		6.5		6.2		6.2		6.2		6.2		6.1		6.25		
Fluoride	0.7	0.2	0.7				0.7		0.7		0.6		0.6		0.7		0.6		0.65714	1.07	1.32
Iron, dissolved			0.069	U	0.069	U	0.069	U	0.069	U	0.069	U	0.069	U	0.069	U	0.069	U	0.0345	0.004	0.017
Iron, total			0.069	U			0.069	U	0.069	U	0.069	U	0.069	U	0.069	U	0.069	U	0.0345	0.01	0.169
Lead, dissolved			0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.0005	0.001	0.002
Lead, total	ND	0.003	0.001	U			0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.0005	0.001	0.002
Lithium, dissolved			0.04408	U	0.04446	U	0.04541	U	0.04646	U	0.04417	U	0.04608	U	0.04246	U	0.04532	U	0.0224		
Lithium, total			0.05143	U			0.04522	U	0.04627	U	0.0457	U	0.04436	U	0.04427	U	0.04704	U	0.02316	0.03314	0.055
Magnesium			20.40975				20.84635		20.4766		20.81502		21.09662		20.24404		20.12109		20.5728	11.4	13.9
Magnesium, dissolved			20.11028		20.53523		20.78437		20.80442		20.81424		21.00295		20.02069		19.80084		20.4841		
Manganese, dissolved			0.00343	U	0.00245	U	0.00232		0.00625	U	0.00698		0.00747		0.006		0.00551		0.00303	0.001	0.002
Manganese, total			0.00514				0.00245		0.00919		0.00845		0.01201		0.00723		0.00723		0.00739	0.001	0.0182

Notes:

Italic indicates Phase II mean value calculated using one-half the value of "U'd results.

Results are in mg/l unless otherwise noted

"U" data qualifier indicates reported value is "non detect" (ND).

"J" data qualifier indicates reported value is estimated.

Table 4.5-1A (continued)
Analysis Results for Water Samples Collected from IWW Ditch

Table 4.5-1A

Parameter	NEIC Sample		Phase II Samples																Mean of Phase II Samples	Well 125	
	NEIC 041W		O307IWA		O307IWD		O307IWE		O307IWF		O307IWG		O307IWH		O307IWN		O307IWP			Range of Values(4/92-12/93)	
	Value Reported	Detection Limit	Value Reported	Data Qualifier	Value Reported	Data Qualifier	Value Reported	Data Qualifier	Value Reported	Data Qualifier	Value Reported	Data Qualifier	Value Reported	Data Qualifier	Value Reported	Data Qualifier	Value Reported	Data Qualifier			Minimum Value
Mercury, dissolved			0.00006	U	0.00006	U	0.0001	U	0.00007	U	0.00006	U	0.00006	U	0.0001	U	0.00009	U	3.8E-05		
Mercury, total	ND	0.0005	0.00011	U			0.00011	U	0.00009	U	0.00013	U	0.00009	U	0.00009	U	0.00011	U	5.2E-05	0.0002	0.00047
Molybdenum, dissolved			0.013	U	0.013	U	0.013	U	0.01395	U	0.013	U	0.01383	U	0.013	U	0.02427	U	0.00732		
Molybdenum, total			0.01803	U			0.01372	U	0.013	U	0.013	U	0.013	U	0.013	U	0.01565	U	0.0071	0.013	0.02
Nickel, dissolved			0.011	U	0.011	U	0.011	U	0.011	U	0.011	U	0.011	U	0.011	U	0.011	U	0.0055		
Nickel, total	ND	0.025	0.011	U			0.011	U	0.011	U	0.011	U	0.011	U	0.011	U	0.011	U	0.0055	0.007	0.02
Nitrate (NO3 as N)			1.25		1.37		1.21		1.21		1.13		1.24		1.26		1.13		1.225	0.47	0.77
Orthophosphate (PO4 as P)	0.58	0.02	0.312		0.48		0.428		0.633		0.385		0.383		0.338		0.3		0.40738	0.02	0.04
pH			8.8		8.9		8.7		8.7		8.8		8.8		8.5		9		8.775	7.27	8.42
Phosphorus, total	0.7	0.02	0.345		0.515		0.495		0.855		0.47		0.44		0.395		0.335		0.48125	0.02	0.34
Potassium			7.96479				8.23679		9.52707		8.12173		8.29607		7.89505		8.045		8.29807	5.9	12.3
Potassium, dissolved			7.81833		7.94387		8.19496		9.61424		8.20541		8.26469		7.79391		7.62304		8.18231		
Redox (mV)			120		77		103		98		81		81		101		101		95.25	40	153
Selenium, dissolved			0.002	U	0.002	U	0.002	U	0.002	U	0.002	U	0.002	U	0.002	U	0.002	U	0.001	0.002	0.002
Selenium, total	ND	0.005	0.002	U			0.002	U	0.002	U	0.002	U	0.002	U	0.002	U	0.002	U	0.001	0.001	0.0031
Silver, dissolved			0.004	U	0.004	U	0.004	U	0.004	U	0.004	U	0.004	U	0.004	U	0.004	U	0.002	0.003	0.005
Silver, total	ND	0.01	0.004	U			0.004	U	0.004	U	0.004	U	0.004	U	0.004	U	0.004	U	0.002	0.002	0.004
Sodium			58.10832				61.15139		54.90327		58.134		59.20499		57.1022		59.18787		58.256	47.6	57
Sodium, dissolved			57.36854		52.80345		60.13637		58.74538		58.89979		59.17864		56.88919		56.71009		57.5914		
Specific conductance, at 25 C (umhos/cm)			1351		592		644		684		200		200		696		712		634.875	414	567
Sulfate			75				75		78		72		80		77		69		75.1429	38	47
Temperature (° C)			22.2		18.5		22.1		23.3		24.2		24.2		18.7		19.9		21.6375	14.8	18.3
Thallium, total	ND	0.01	0.001	U			0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.001	U	0.0005	0.0005	0.004
Thallium, dissolved			0.001	U	0.001	U	0.001	U	0.001	U	0.00151		0.001	U	0.001	U	0.001	U	0.00063		
Total dissolved solids			490		400		490		430		410		440		460		1810		616.25	300	390
Total suspended solids			4	U	4	U	4	U	6	U	4	U	4	U	4	U	4	U	4.25		
Vanadium, dissolved			0.002	U	0.00212	U	0.00359	U	0.00429	U	0.00249	U	0.00475	U	0.002	U	0.00304	U	0.00152	0.0045	0.007
Vanadium, total			0.00442	U			0.00242	U	0.00412	U	0.00525	U	0.0023	U	0.00377	U	0.00515	U	0.00196	0.002	0.12
Zinc, dissolved			0.01866	U	0.01232	U	0.00999	U	0.00803	U	0.01408	U	0.00723	U	0.00677	U	0.00845	U	0.00535	0.003	0.0157
Zinc, total	ND	0.02	0.01503	U			0.01379	U	0.03087	U	0.01137	U	0.01964	U	0.01938	U	0.01835	U	0.00917	0.0051	0.065
Radiological Activities (pCi/l)																					
Gross alpha			2.12	UJ	2.3	UJ	0.05	U	0.74	U	1.71	U	3.42		-0.69	U	-0.13	U		1.26	3.7
Gross beta			4.83	U	5.17	U	4.94	U	5.71	U	4.62	U	5.53	U	5.1	U	6.09	U		4.44	10.2
Radium-226			0.89	U	0.31	U	0.46	U	0.79	U	0.18	U	0.63	U	0	U	0.18	U		0.16	1.5
Radium-228			-0.3	U	-2.9	U	-1.5	U	-0.3	U	6.5		-1.1	U	0	U	-0.7	U		0.4	1.7

Notes:

Italic indicates Phase II mean value calculated using one-half the value of "U'd results.

Results are in mg/l unless otherwise noted

"U" data qualifier indicates reported value is "non detect" (ND).

"J" data qualifier indicates reported value is estimated.

TABLE 4.5-2
MEAN METAL CONCENTRATIONS IN SPRING WATER (mg/l)

TABLE 4.5-2

Parameter	SPRING GROUP													Representative Groundwater Concentrations
	I - Batiste System		II - Swanson Road System	III - East Side System		IV - Papoose System								
	Batiste Spring	Batiste Springs Drainage	Swanson Road Spring	Springs near STP	Spring-fed Pond at FMC Park	Papoose Spring	Papoose Springs Drainage	Papoose Springs Drainage Channel	Papoose Springs Drainage Channel	Papoose Springs Discharge	Siphon Road Spring	Twenty Springs (East)		
	SW14	SW11	SW15	SW13	SW09	SW07	SW06	SW5F(a)	SW5E(a)	SW05	SW04	SW02		
Arsenic														
Total	0.015	0.006	0.007	0.003	0.003	0.003	0.004	0.002	0.004	0.004	0.003	0.004	0.0104 to 0.018	
Dissolved	0.032	0.003	0.010	0.003	0.004	0.005	0.003	0.002	0.003	0.003	0.003	0.002		
Barium														
Total	0.083	0.107	0.123	0.114	0.104	0.095	0.094	0.077	0.074	0.103	0.064	0.760	0.12 to 0.23	
Dissolved	0.061	0.086	0.114	0.112	0.089	0.075	0.081	0.079	0.076	0.089	0.062	0.063		
Boron														
Total	0.19	0.21	0.28	0.24	0.10	0.19	0.10	ND	ND	0.09	0.08	0.10	0.25 to 0.308	
Dissolved	0.18	0.19	0.21	0.22	0.10	0.23	0.20	ND	ND	0.12	0.15	0.11		
Lithium														
Total	0.051	0.034	0.044	0.036	0.023	0.031	0.033	ND	ND	0.036	0.038	0.037	0.0165 to 0.0610	
Dissolved	0.053	0.038	0.041	0.042	0.023	0.033	0.037	0.024	0.027	0.035	0.039	0.039		
Vanadium														
Total	0.018	0.029	0.026	0.013	0.030	0.032	0.023	ND	ND	0.024	0.006	0.006	0.0745 to 0.199	
Dissolved	0.004	0.029	0.034	0.003	0.026	0.003	0.057	ND	ND	0.067	ND	ND		

Notes: (a) Sampled only in April 1993.

NA = not analyzed.

ND = not detected/below detection limit.

Reference:

Means for RI data calculated from four rounds of sampling: July and October 1992, and February and April 1993. Samples with results reported as below detection not used in calculation of individual station means. Station mean reported as ND if all sample results below detection.

TABLE 4.5-3
MEAN METAL CONCENTRATIONS IN RIVER WATER (mg/l)

TABLE 4.5-3

Parameter (units in mg/l)	Losing River Stations										River Station SW17	Gaining River Stations								Representative Groundwater Concentrations
	SW25	SW24	SW23	SW22	SW21	SW20	SW19	SW18 ^(a)	SW16	Group Mean	SW17	SW12E ^(b)	SW12	SW10	SW08	SW7E ^(b)	SW03	SW01	Group Mean	
Aluminum																				
Total (4 events)	0.35	0.49	0.66	0.85	0.83	0.44	0.51	0.07	0.53	0.52	1.02	1.80	0.50	0.59	0.76	1.32	1.07	0.57	0.94	0.57 to 0.95
Total (3 events)	0.17	0.16	0.22	0.30	0.30	0.12	0.17	0.07	0.13	0.18	ND	NS	0.10	0.17	0.44	NS	ND	0.02	0.18	
Total (Apr-93)	0.90	1.47	1.55	1.40	1.36	1.08	1.54	NS	1.73	1.38	1.02	1.80	0.90	1.00	1.09	1.32	1.07	1.13	1.19	
Dissolved	ND	0.08	ND	0.02	ND	ND	0.20	ND	0.10	0.10	ND	ND	0.10	0.15	0.16	ND	ND	0.03	0.11	
Arsenic																				
Total	0.004	0.005	0.004	0.006	0.005	0.007	0.007	0.007	0.007	0.006	0.006	0.005	0.005	0.003	0.003	0.003	0.006	0.005	0.004	0.0104 to 0.018
Dissolved	0.004	0.004	0.003	0.003	0.003	0.006	0.003	ND	0.003	0.004	0.003	0.005	0.006	0.004	0.006	0.004	0.003	0.003	0.004	
Barium																				
Total	0.104	0.101	0.103	0.100	0.099	0.102	0.104	0.109	0.090	0.101	0.081	0.096	0.124	0.119	0.116	0.093	0.090	0.092	0.104	0.12 to 0.23
Dissolved	0.092	0.094	0.094	0.093	0.088	0.099	0.104	0.108	0.095	0.096	0.096	0.076	0.096	0.100	0.102	0.078	0.082	0.137	0.096	
Boron																				
Total	0.33	0.22	0.23	0.12	0.11	0.13	0.25	0.12	0.19	0.19	0.38	ND	0.27	0.19	0.18	ND	0.14	0.16	0.19	0.25 to 0.308
Dissolved	0.17	0.31	0.14	0.15	0.18	0.16	0.20	0.13	0.19	0.18	0.23	ND	0.20	0.20	0.17	ND	0.14	0.15	0.17	
Copper																				
Total	0.009	0.003	ND	ND	ND	ND	ND	ND	0.005	0.006	0.015	ND	0.007	ND	ND	ND	0.003	ND	ND	0.0085 to 0.013
Dissolved	0.003	0.003	ND	0.003	ND	ND	0.022	ND	0.005	0.007	0.011	ND	0.003	0.004	0.004	ND	0.012	ND	0.006	

Notes: (a) SW18 sampled only during October 1992.

(b) Sampled only in April 1993.

NA = not analyzed.

ND = Not detected/below detection limit.

NS = Not sampled.

Reference: Means for RI data calculated from four rounds of sampling: July and October 1992, and February and April 1993. Samples with results reported as below detection not used in calculation of individual station means. Station mean reported as ND if all sample results below detection. Group mean reported as ND if 70 percent (All River and Losing Reach) to 60 percent (Gaining Reach) of station means below detection limits.

TABLE 4.5-3 (continued)
MEAN METAL CONCENTRATIONS RIVER WATER (mg/l)

TABLE 4.5-3

Parameter (units in mg/l)	Losing River Stations										River Station SW17	Gaining River Stations								Representative Groundwater Concentrations
	SW25	SW24	SW23	SW22	SW21	SW20	SW19	SW18 ^(a)	SW16	Group Mean	SW17	SW12E ^(b)	SW12	SW10	SW08	SW7E ^(b)	SW03	SW01	Group Mean	
Iron																				
Total (4 events)	0.50	0.54	0.63	0.43	0.54	0.33	0.61	0.07	0.54	0.47	0.41	1.73	0.38	0.31	0.34	1.32	0.42	0.40	0.70	0.7690 to 1.074
Total (3 events)	0.06	0.09	0.14	0.12	0.15	0.15	0.13	0.07	0.18	0.12	0.09	NS	0.08	0.07	0.11	NS	0.04	0.04	0.07	
Total (Apr-93)	0.94	1.46	1.61	1.37	1.34	1.13	1.56	NS	1.61	1.38	1.05	1.73	0.97	1.03	1.03	1.32	1.17	1.13	1.20	
Dissolved	0.06	0.09	ND	0.02	0.09	0.02	0.05	ND	0.03	0.05	0.09	ND	0.06	0.05	0.04	ND	0.04	0.04	0.05	
Lithium																				
Total	0.058	0.058	0.055	0.052	0.057	0.053	0.051	0.056	0.049	0.054	0.049	0.023	0.039	0.041	0.037	ND	0.039	0.043	0.037	0.0165 to 0.061
Dissolved	0.058	0.057	0.056	0.046	0.043	0.054	0.090	0.056	0.041	0.056	0.042	ND	0.045	0.041	0.037	0.021	0.041	0.038	0.037	
Manganese																				
Total	0.020	0.023	0.025	0.021	0.027	0.020	0.027	0.008	0.024	0.022	0.023	0.057	0.018	0.018	0.017	0.046	0.019	0.016	0.027	0.0201 to 0.1097
Dissolved	0.004	0.032	ND	0.002	ND	0.004	0.035	ND	0.004	0.013	0.005	0.001	0.005	0.004	0.004	0.001	0.002	0.004	0.003	
Vanadium																				
Total	0.013	0.005	0.012	0.003	0.022	0.004	ND	0.004	0.037	0.013	0.027	ND	0.019	0.023	0.031	ND	0.005	0.017	0.019	0.0745 to 0.1987
Dissolved	ND	ND	ND	ND	ND	0.002	0.016	ND	0.057	0.025	0.012	ND	0.070	0.047	0.079	ND	0.003	0.019	0.044	

Notes: (a) SW18 sampled only during October 1992.

(b) Sampled only in April 1993.

NA = not analyzed.

ND = Not detected/below detection limit.

NS = Not sampled.

Reference: Means for RI data calculated from four rounds of sampling: July and October 1992, and February and April 1993. Samples with results reported as below detection not used in calculation of individual station means. Station mean reported as ND if all sample results below detection. Group mean reported as ND if 70 percent (All River and Losing Reach) to 60 percent (Gaining Reach) of station means below detection limits.

TABLE 4.5-4
MEAN CONCENTRATIONS OF COMMON IONS, PHYSICAL PARAMETERS, AND NUTRIENTS AND FLUORIDE IN SPRING WATER

Parameter	I - Batiste System			II - Swanson Road System		III - East Side System			IV - Papoose System								Representative Groundwater Concentration
	System Mean	Batiste Spring	Batiste Springs Drainage	System Mean	Swanson Road Spring	System Mean	Springs near STP	Spring-fed Pond at FMC Park	System Mean	Papoose Spring	Papoose Springs Drainage	Papoose Spring Drainage Channel		Papoose Springs Discharge	Siphon Road Spring	Twenty Springs (East)	
	1978-80 ^(a)	1992-93 ^(a)	1992-93 ^(a)	1978-80 ^(a)	1992-93 ^(a)	1978-80 ^(a)	1992-93 ^(a)	1992-93 ^(a)	1978-80 ^(a)	1992-93 ^(a)	1992-93 ^(a)	April 1993 ^(b)		1992-93 ^(a)	1992-93 ^(a)	1992-93 ^(a)	
	Perry et al. (1990)	SW14	SW11	Perry et al. (1990)	SW15	Perry et al. (1990)	SW13	SW09	Perry et al. (1990)	SW07	SW06	SW5E	SW5F	SW05	SW04	SW02	
Common Ions																	
Alkalinity, bicarbonate, mg/l	233.3	211	222	290.2	286	264.4	278	357	201.1	252	205	190	204	211	181	186	171 to 233
Calcium, mg/l	101.3	69.5 (78.7)	58.0 (64.1)	79.5	92.7 (98.5)	70.2	64.6 (67.9)	59.7 (63.7)	60.5	57.1 (61.2)	57.5 (61.4)	59.4	57.8	59.0 (63.3)	54.0 (58.1)	54.4 (58.4)	68.75 to 101
Chloride, mg/l	55.7	40.2	27.9	40.3	49.3	26.8	27.5	21.7	20.9	17.9	18.8	17.0	20.0	26.5	15.2	15.5	52.4 to 55.7
Magnesium, mg/l	34.4	26.6	22.5	27.8	33.5	27	27.5	25.8	21.4	20.1	19.9	19.8	20.9	22.4	16.9	16.9	19.2 to 34.4
Potassium, mg/l	9.5	8.79	5.84	6.7	7.37	6.2	6.97	6.50	5.4	4.15	4.26	3.66	3.91	5.28	3.54	3.53	7.34 to 9.5
Sodium, mg/l	53.6	52.9	42.7	41.4	55.7	36.1	54.4	37.2	23.4	23.8	23.4	21.0	23.6	31.2	20.3	20.6	27.5 to 53.6
Sulfate, mg/l	~150	113	51	~40	104	~55	55	45	~45	38	39	39	39	43	38	39	43.4 to ~150
Physical Parameters																	
Specific conductance, μ mhos/cm	946.3	773	609	732.5	907.7	638.5	747.0	658.5	515.7	495.7	495.7	519	535	565.3	469.0	458.3	569 to 946
pH, units		7.3	7.8		7.3		7.3	7.5		7.8	7.8	8.0	8.2	7.8	7.7	7.8	7.3 to 8.2
Temperature, °C		13.9	14.4		13.6		17.1	17.5		14.3	14.6	12.4	13.4	14.0	16.3	16.4	13.6 to 17.5
Total dissolved solids, mg/l		465	360		540		415	350		300	293	260	330	320	290	313	260 to 465
Total suspended solids, mg/l		ND	9		ND		6.0	4.0		ND	ND	NM	NM	ND	22.0	21.0	4.0 to 9
Nutrients and Fluoride																	
Ammonia (NH ₃ as N), mg/l	6.19	ND	0.3	0.05	0.4	0.22	ND	ND	0.08	0.5	ND	ND	ND	0.4	ND	ND	0.3 to 6.19
Nitrate (NO ₃ as N), mg/l	5.58	4.44	1.99	2.54	2.64	2.1	3.41	2.30	1.43	2.98	2.14	2.47	2.56	2.15	1.40	1.47	1.60 to 5.58
Orthophosphate (PO ₄ as P), mg/l	1.9	2.36	0.59	0.04	0.99	0.1	0.04	0.03	0.03	0.03	0.03	0.03	ND	0.24	0.04	0.03	0.06 to 2.36
Phosphorus, total, mg/l	2.06	2.71	0.48	0.07	1.05	0.14	0.05	0.04	0.07	ND	0.04	0.05	ND	0.22	0.07	0.09	0.15 to 2.71
Fluoride, mg/l	0.44	0.6	0.6	0.3	0.5	0.42	0.3	0.4	1.32	0.5	0.6	0.6	0.5	0.5	0.8	0.7	0.41 to 1.32
Dissolved oxygen, mg/l		6.5	9.9		4.7		8.0	9.9		8.0	8.9	8.2	9.4	7.5	6.5	7.7	6.5 to 9.9

Notes: (a) Sampling dates.
(b) Sampling date was April 1993. Numbers indicate one round of sampling, not a mean.
ND = not detected/below detection limit.

Reference: Means for RI data calculated from four rounds of sampling: July and October 1992, and February and April 1993; except calcium value in parentheses calculated from October 1992, February 1993, and April 1993 data only. Samples with results reported as below detection not used in calculation of means. Mean reported as ND if all sample results below detection.

TABLE 4.5-5
MEAN CONCENTRATIONS OF COMMON IONS, PHYSICAL PARAMETERS, AND NUTRIENTS AND FLUORIDE IN RIVER WATER

TABLE 4.5

Parameter	Losing River Stations										River Station SW17	Gaining River Stations									Representative Groundwater Concentration
	SW25	SW24	SW23	SW22	SW21	SW20	SW19	SW18 ^(a)	SW16	Group Mean	SW17	SW12E	SW12	SW10	SW08	SW7E	SW03	SW01	Group Mean		
Common Ions																					
Alkalinity, bicarbonate, mg/l	250	253	248	248	231	257	249	284	246	252	188	22.4	268	251	251	21.2	246	241	251	171 to 307	
Calcium, mg/l	62.0 (72.3)	61.6 (71.9)	62.9 (72.6)	53.4 (65.2)	55.8 (65.8)	65.9 (73.7)	65.7 (75.0)	69.8 (69.8)	65.5 (72.1)	62.5 (70.9)	62.2 (67.6)	69.1	70.2 (75.3)	66.1 (72.9)	65.5 (70.3)	63.9	66.0 (70.1)	65.1 (69.1)	66.6 (71.5)	68.75 to 97.7	
Chloride, mg/l	46.7	48.1	47.7	45.5	42.8	46.5	45.7	56.3	46.5	47.3	54.6	32.0	49.0	43.5	38.5	33.0	40.9	39.6	42.3	52.4 to 192.9	
Magnesium, mg/l	32.4	31.1	32.2	30.6	28.9	32.7	32.0	37.2	30.5	32.0	22.6	25.3	30.6	27.9	27.4	23.5	26.8	26.1	27.8	19.2 to 35.9	
Potassium, mg/l	9.57	8.95	9.23	8.77	8.50	9.43	8.73	10.80	8.45	9.16	8.36	5.66	9.12	7.92	7.49	5.62	7.50	7.21	7.85	7.34 to 12.7	
Sodium, mg/l	41.0	41.6	41.7	40.2	43.5	41.5	39.7	48.1	41.1	42.1	47.7	29.8	55.1	48.6	42.7	32.3	47.1	51.2	49.0	27.5 to 74.3	
Sulfate, mg/l	50	38	42	42	38	43	41	46	45	42.9	65	41	70	60	54	49	58	57	59.8	43.4 to 72.6	
Physical Parameters																					
Specific conductance, μ mhos/cm	730.3	707.3	702.7	677.3	647.7	696.0	631.0	875.0	678.7	705	738.0	614	775.7	667.3	685.3	617	673.7	642.3	689	569 to 1,136	
pH, units	8.5	8.4	8.4	8.4	8.5	8.7	8.3	8.6	8.2	8.5	8.5	8.0	7.5	7.7	7.9	7.9	7.6	7.7	7.7		
Temperature, °C	17.0	15.6	15.6	15.9	17.2	14.7	17.3	12.4	13.8	15.5	17.6	11.5	14.9	14.1	16.2	10.9	16.3	15.8	15.5		
Total dissolved solids, mg/l	403	393	380	347	300	405	380	NM	400	376	403	350	460	420	425	340	410	393	422		
Total suspended solids, mg/l	9.0	12.0	14.0	17.0	52.0	22.0	15.0	NM	9.0	19	5.0	NM	6.5	6.0	28.0	NM	4.0	4.0	10		
Nutrients																					
Ammonia (NH ₃ as N), mg/l	ND	0.3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	3.4	1.2	0.9	ND	0.8	0.8	1.4	0.5	
Nitrate (NO ₃ as N), mg/l	0.64	0.68	0.59	0.65	0.42	0.60	0.66	0.31	0.82	0.60	1.29	1.23	2.09	1.91	1.93	0.82	2.14	2.26	2.07	1.60 to 5.52	
Orthophosphate (PO ₄ as P), mg/l	0.03	0.03	0.05	0.03	0.03	0.03	0.03	ND	0.07	0.04	0.32	0.34	0.97	0.45	0.39	0.38	0.48	0.45	0.55	0.06 to 0.27	
Phosphorus, total, mg/l	0.09	0.09	0.09	0.08	0.16	0.06	0.13	0.02	0.15	0.10	0.64	0.47	1.05	0.49	0.43	0.48	0.52	0.45	0.59	0.15 to 0.33	
Fluoride, mg/l	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.7	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.41 to 0.80	
Dissolved oxygen, mg/l	12.7	9.1	9.1	9.2	9.9	9.3	9.5	10.4	9.4	9.8	9.3	7.3	8.5	7.4	7.8	10.2	7.4	9.2	8.1		

Notes: (a) Sampled only in October 1992.

ND = Not detected/below detection limit.

NM = Not measured.

Reference: Means for RI data calculated from four rounds of sampling: July and October 1992, and February and April 1993; except calcium value in parentheses calculated from October 1992, and February and April 1993 data only. Samples with results reported as below detection not used in calculation of means. Mean reported as ND if all sample results below detection.

TABLE 4.5-6
RADIOLOGICAL ACTIVITIES IN SPRING WATER (pCi/l)

TABLE 4.5-6

Parameter (pCi/l)	SPRING GROUP											
	I - Batiste System		II - Swanson Road System	III - East Side System		IV - Papoose System						
	Batiste Spring	Batiste Springs Drainage	Swanson Road Spring	Springs near STP	Spring-fed Pond at FMC Park	Papoose Spring	Papoose Springs Drainage	Papoose Springs Drainage Channels		Papoose Springs Discharge	Siphon Road Spring	Twenty Springs (East)
	SW14	SW11	SW15	SW13	SW09	SW07	SW06	SW05F	SW05E	SW05	SW04	SW02
Gross alpha												
Jul-92	2.30±1.50	2.20±1.40	ND	ND	2.60±1.60	2.00±1.30	2.10±1.30	NS	NS	ND	ND	ND
Oct-92	2.22±0.50	2.31±0.50	ND	2.66±0.57	2.66±0.50	ND	ND	NS	NS	ND	ND	ND
Feb-93	2.97±1.67	2.32±1.89	3.51±1.62	2.86±1.54	5.55±1.73	ND	ND	NS	NS	2.73±1.64	ND	3.26±1.71
Apr-93	ND	3.50±1.17	2.21±0.91	2.90±1.27	3.11±0.93	8.84±2.30	6.58±1.51	ND	2.06±0.84	2.59±1.15	ND	ND
Gross beta												
Jul-92	6.10±1.70	5.50±1.70	8.00±1.80	7.10±1.50	7.30±1.80	4.20±1.60	3.90±1.60	NS	NS	4.40±1.60	3.30±1.50	4.90±1.60
Oct-92	7.63±0.63	5.56±0.58	6.10±1.70	6.42±0.58	5.63±0.52	3.20±1.70	2.70±1.50	NS	NS	5.50±1.90	2.80±1.60	4.60±1.80
Feb-93	6.41±1.00	4.98±1.11	7.37±1.18	5.14±1.15	7.10±1.01	3.11±0.98	2.89±0.97	NS	NS	4.83±1.17	3.39±0.87	2.83±0.83
Apr-93	11.00±3.55	7.23±1.46	ND	6.43±1.41	5.69±1.33	±13.70±4.70	20.40±4.59	3.96±1.40	6.65±1.48	7.99±3.47	4.92±1.30	6.93±3.52
Radium-226												
Jul-92	ND	ND	ND	ND	ND	1.72±0.25	ND	NS	NS	ND	ND	ND
Oct-92	ND	2.60±0.40	1.82±0.25	ND	ND	ND	ND	NS	NS	5.20±0.26	ND	ND
Feb-93	ND	ND	1.50±0.62	ND	ND	1.93±0.33	ND	NS	NS	1.72±0.58	ND	ND
Apr-93	ND	ND	ND	ND	ND	1.40±0.38	ND	ND	ND	ND	ND	ND
Radium-228												
Jul-92	ND	ND	1.3±0.5	ND	ND	ND	ND	NS	NS	ND	ND	ND
Oct-92	ND	ND	2.2±1.0	ND	ND	1.4±0.9	1.7±0.9	NS	NS	ND	3.5±0.9	3.2±1.0
Feb-93	ND	ND	ND	1.4±0.9	ND	1±0.8	ND	NS	NS	ND	ND	5.3±1.2
Apr-93	ND	ND	ND	ND	ND	ND	ND	2.1±0.8	2.8±1.2	1.1±0.8	ND	ND
Uranium-233/234												
Jul-92	NA	NA	NA	NA	NA	NA	NA	NS	NS	NA	NA	NA
Oct-92	NA	NA	NA	NA	NA	NA	NA	NS	NS	NA	NA	NA
Feb-93	1.08±0.27	NA	NA	1.67±0.52	NA	NA	NA	NS	NS	1.19±0.32	NA	NA
Apr-93	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Notes: NA = Not analyzed
 ND = Not detected/below detection limit.
 NS = Not sampled.

TABLE 4.5-7
RADIOLOGICAL ACTIVITIES IN RIVER WATER (pCi/l)

TABLE 4.5-7

Parameter (pCi/l)	Losing River Stations									River Station SW17	Gaining River Stations						
	SW25	SW24	SW23	SW22	SW21	SW20	SW19	SW18	SW16		SW12E	SW12	SW10	SW08	SW07E	SW03	SW01
Gross alpha																	
Jul-92	ND	ND	ND	2.60±1.60	ND	ND	2.70±1.70	NS	3.00±1.60	ND	NS	ND	2.20±1.40	2.50±1.50	NS	2.50±1.70	2.10±1.60
Oct-92	ND	ND	ND	ND	ND	2±1.5	2.70±1.80	ND	ND	ND	NS	2.86±0.69	2.45±0.63	4.27±0.60	NS	ND	ND
Feb-93	4.93±2.00	2.03±1.52	2.60±1.84	3.31±1.58	NS	3.60±1.96	4.34±2.04	NS	2.61±1.94	3.69±1.54	NS	3.78±1.21	3.04±1.49	3.22±1.34	NS	3.60±1.55	3.22±1.19
Apr-93	2.20±0.94	ND	6.55±1.48	2.59±1.11	ND	ND	2.37±0.90	NS	ND	ND	2.53±1.13	2.04±1.36	10.80±2.10	3.09±0.93	2.30±1.24	2.63±0.87	2.87±1.02
Gross beta																	
Jul-92	8.10±1.30	6.90±1.80	8.40±1.30	9.60±1.30	6.60±1.20	7.00±1.20	9.10±1.30	NS	6.90±1.80	9.90±1.90	NS	7.30±1.80	4.00±1.60	5.70±1.70	NS	6.40±1.80	12.00±2.00
Oct-92	9.80±1.80	7.10±1.70	9.30±1.80	7.90±1.70	9.30±1.80	7.7±1.7	9.80±1.80	9.40±1.80	8.40±1.80	5.80±1.60	NS	9.04±0.67	10.70±0.78	9.47±0.66	NS	6.60±2.00	4.20±1.60
Feb-93	8.67±1.23	9.05±1.24	7.08±1.37	7.43±1.23	NS	8.21±1.42	7.89±1.41	NS	7.41±1.22	7.68±1.15	NS	8.05±1.02	7.97±1.28	7.40±1.02	NS	8.80±1.16	7.10±0.81
Apr-93	ND	ND	13.80±4.31	ND	ND	11.10±3.87	ND	NS	ND	ND	3.41±1.74	10.10±4.64	8.71±3.57	8.39±1.42	5.20±1.45	6.80±1.44	5.96±1.35
Radium-226																	
Jul-92	ND	ND	ND	ND	ND	ND	ND	NS	ND	ND	NS	ND	ND	ND	NS	3.11±0.27	ND
Oct-92	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	NS	ND	ND	ND	NS	ND	ND
Feb-93	ND	2.91±0.78	ND	ND	NS	ND	ND	NS	ND	1.19±0.50	NS	ND	ND	ND	NS	1.15±0.49	ND
Apr-93	1.31±0.38	ND	ND	ND	ND	ND	ND	NS	ND	1.14±0.36	ND	ND	ND	ND	ND	ND	1.00±0.34
Radium-228																	
Jul-92	ND	ND	7.5±0.8	ND	1.2±0.5	1.9±0.9	ND	NS	ND	ND	NS	ND	ND	ND	NS	ND	ND
Oct-92	2.1±1.1	2.3±1.0	3.2±1.0	ND	1.5±0.9	1.7±0.9	2.0±1.0	2.2±1.0	ND	ND	NS	ND	ND	ND	NS	5.9±2.2	2.1±0.9
Feb-93	ND	ND	ND	ND	NS	3.4±0.9	ND	NS	ND	ND	NS	ND	ND	ND	NS	ND	ND
Apr-93	3.3±0.8	ND	1.6±0.8	ND	1.2±0.7	ND	ND	NS	1.6±0.8	ND	2.0±0.8	1.1±0.8	ND	ND	1.2±0.9	1.8±1.0	ND
Uranium-233/234																	
Jul-92	NA	NA	NA	NA	NA	NA	NA	NS	NA	NA	NS	NA	ND	NA	NS	NA	NA
Oct-92	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NS	NA	NA	NA	NS	NA	NA
Feb-93	1.26±0.29	1.39±0.29	NA	1.40±0.35	NS	NA	NA	NS	NA	1.12±0.47	NS	NA	1.23±0.30	NA	NS	NA	1.36±0.51
Apr-93	NA	NA	NA	NA	NA	NA	NA	NS	NA	NA	NA	NA	NA	NA	NA	NA	NA

Notes: NA = Not analyzed.

ND = Not detected/below detection limit.

NS = Not sampled.

Section 4 Nature and Extent of Constituents of Potential Concern

TABLE 4.5-8
PERCENTAGE OF GROSS BETA ACCOUNTED FOR BY POTASSIUM-40

Station ID	Sampling Round			
	Jul-92	Oct-92	Feb-93	Apr-93
1	52.5	162.3	96.0	77.8
2	60.9	68.5	108.4	40.7
3	102.5	107.1	81.3	69.9
4	92.9	109.5	93.0	56.8
5	83.3	106.9	75.9	59.9
5E	NS	NS	NS	46.9
5F	NS	NS	NS	84.1
6	107.0	132.5	123.8	15.6
7	89.3	114.5	115.1	23.1
8	95.7	81.0	99.0	60.5
9	82.9	93.8	74.4	97.4
10	117.2	82.8	103.7	59.4
11	102.2	75.1	94.1	74.9
12	95.7	88.6	93.1	85.2
12E	NS	NS	NS	141.4
13	90.0	95.6	111.1	85.8
14	92.2	75.9	90.4	115.9
15	82.0	111.7	84.4	ND
16	98.8	106.5	112.7	ND
17	83.5	123.4	102.1	ND
18	NS	97.9	NS	NS
19	78.6	95.6	110.1	ND
20	126.6	102.9/106.7	103.8	39.3
21	118.8	100.8	NS	ND
22	82.5	115.4	113.5	ND
23	109.5	98.9	121.5	32.2
24	125.9	127.2	92.3	ND
25	126.2	94.8	99.3	ND

Notes: All results given in %.
 ND = Not detected/below detection limit.
 NS = Not sampled.

TABLE 4.5-9
METAL CONCENTRATION IN RIVER SEDIMENTS

TABLE 4.5-9

Parameter (mg/kg)	Upstream River Stations											River Station SW17	Downstream River Stations										Background Soil EPA
	SD25 (clayey sand)	SD24 (clay)	SD23 (clay)	SDA2 (silty clay)	SDA1 (silty clay)	SD22 (silty clay)	SD21 (silt and sand)	SD20 (sand w/ silt and gravel)	SD19 (silty clay)	SD18 (sand w/ gravel)	SD16 (silty clay)	SD17 (sandy clay)	SD12 (sand)	SD10 (fine sand)	SD08 (loam)	SD03 (loam, sand, and gravel)	SD01 (loam)	SDB1 (silty clay)	SDC1 (silt w/ fine sands)	SDC2 (silt w/ fine sands)	SDC4 (silt w/ fine sands)		
Aluminum, total	4850	10400	11400	11200 J	14600 J	5230	4450	2100	8560	3690	9010	7830	3810	2500	5200	3670	4260	16200 J	5320 J	8760 J	6790 J	13,900	
Arsenic, total	4.1	4.1	3.5	5.5	5.7	3.4	3.6	ND	2.7	8.4	3.0	3.7	3.7	2.4	9.9	3.1	4.4	6.2	6.5	4.6	6.1	7.7	
Barium, total	174	122	113	123 J	145 J	112	87.3	130	144	109 J	123 J	165 J	108 J	68.7 J	95.8 J	71.7 J	95.6 J	183 J	118 J	122 J	106 J	188	
Beryllium, total	0.21	0.65	0.41	0.62	0.81	0.10	ND	ND	ND	ND	0.80	0.73	1.10	0.35	1.50	ND	0.79	0.89	0.32	0.50	0.32	1	
Boron, total	4.1	4.3	6	12.2	13.2	5.2	3	3.5	4.6	2.5	2.7	3.3	3.2	4.0	4.9	4.2	5.7	15.2	ND	13.1	12.6	12.8	
Cadmium, total	ND	ND	ND	ND	ND	ND	ND	ND	1.6	ND	ND	22.2	ND	ND	ND	ND	ND	ND	ND	1	0.95	1.9	
Chromium, total	9.2	18.1	19.4	17.2	18.9	10.6	12.4	5.6	22.4	9.6 J	23.8 J	80.8 J	8.2 J	12.1 J	16.6 J	12.4 J	9.7 J	25.0	11.7	19.0	16.2	27.5	
Cobalt, total	5.5	4.5	5.8	5.2	6.4	3.7	4.9	4.4	4.8	ND	ND	ND	ND	ND	ND	ND	ND	6.4	3.6	4.4	3.1	7.6	
Copper, total	12.1	11.6	12.8	11.6	14.8	10.6	10.8	9.8	12.7	7.1 J	30.8 J	85.9 J	9.6 J	4.8 J	9.2 J	6.4 J	8.3 J	25.5	6.3	12.9	9.7	12.6	
Iron, total	7730	10400	11400	10900 J	14400 J	7890	7850	4970	8530	5270 J	10100 J	7770 J	6090 J	5130 J	7020 J	7280 J	6530 J	16100 J	6010 J	8910 J	6920 J		
Lead, total	26.0 J	51.6 J	71.9 J	12.1	13.9	13.8	12.8	61.0 J	38.6 J	12.4 J	17.6 J	21.3 J	9.8 J	6.7 J	19.2 J	9.2 J	22.9 J	30.9	8.1	9.2	8.9	29.1	
Lithium, total	6.7	12.4	14.1	12.5 J	15.8 J	7.2	6.0	3.7	10.6	4.5	9.2	7.2	5.0	4.0	6.7	3.8	5.7	21.8 J	7.7 J	12.1 J	8.3 J	16.1	
Manganese, total	522 J	365 J	216 J	229 J	399 J	277 J	300 J	342 J	237 J	170 J	163 J	1210 J	425 J	99.9 J	97.6 J	136 J	120 J	238 J	202 J	161 J	140 J	482	
Mercury, total	ND	ND	ND	ND	0.55	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	1.1	ND	ND	ND	0.16	
Molybdenum, total	ND	ND	ND	ND	ND	ND	2.5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	2.15	
Nickel, total	6.7	7.6	7.3	11.8	13.9	5.9	4.0	2.7	6.4	4.8	6.9	12.1	4.8	ND	6.5	4.8	ND	16.1	7.3	9.7	7.8	15.5	
Selenium, total	0.72	0.54	ND	ND	ND	ND	ND	ND	1.20	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	1.36	
Silver, total	ND	ND	ND	ND	ND	ND	ND	3.0	ND	ND	1.7	4	2.2	1.4	1.2	0.44	2.1	ND	1.3	ND	ND	1.9	
Thallium, total	ND	ND	ND	ND	ND	0.14 J	ND	0.17	0.18	0.30 J	0.73 J	0.53 J	0.14 J	0.2 J	0.24 J	R	0.28 J	ND	ND	ND	ND	0.27	
Vanadium, total	32.1	28.8	30.7	18.1	21.4	25.1	23.1	24.2	32.3	19.9	25.3	87.8	18.8	15.6	30.6	14.6	21.9	23.9	10.5	17.5	13.9	45.4	
Zinc, total	24.3	42.9	48.1	50.3 J	55.3 J	32.7	40.1	ND	50.3	ND	56.9 J	251.0 J	27.0 J	27.4 J	39.4 J	31.8 J	25.6 J	97.1 J	30.2 J	50.1 J	41.3 J	52.8	

Notes: Units in mg/kg.
J = Estimated value.
ND = Not detected/below detection limit.
R = Rejected value.

TABLE 4.5-10
CONCENTRATIONS OF NUTRIENTS AND OTHER PARAMETERS IN RIVER SEDIMENTS

TABLE 4.5-10

PARAMETER	UPSTREAM RIVER STATIONS											RIVER STATION SW17	DOWNSTREAM RIVER STATIONS									
	SD25 (clayey sand)	SD24 (clay)	SD23 (clay)	SDA2 (silty clay)	SDA1 (silty clay)	SD22 (silty clay)	SD21 (silt and sand)	SD20 (sand w/ silt and gravel)	SD19 (silty clay)	SD18 (sand w/ gravel)	SD16 (silty clay)		SD17 (sandy clay)	SD12 (sand)	SD10 (fine sand)	SD08 (loam)	SD03 (loam, sand, and gravel)	SD01 (loam)	SDB1 (silty clay)	SDC1 (silt w/ fine sands)	SDC2 (silt w/ fine sands)	SDC4 (silt w/ fine sands)
Calcium	NA	NA	NA	49100 J	36500 J	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	69300 J	166000 J	88500 J	93200 J	
Magnesium	NA	NA	NA	5020 J	5510 J	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	8370 J	5610 J	5830 J	4920 J	
Orthophosphate (PO4 as P)	2.0	2.5	0.9	3.3	6.1	0.7	0.6	0.5	0.4	0.9	ND	1.7	1.2	2.4	ND	4.0	0.7	10.7	5.6	2.1	2.8	
Total Phosphorus	158	375	314	521	531	369	231	204	640	471	554	5340	479	7150	577	227	1310	493	1160	707	1060	
Fluoride	193	241	1300	390	460	500	198	149	338	240	273	3080	189	420	237	220	443	505	550	410	340	
pH	7.2	7.6	8.1	7.58	7.66	7.8	8.1	8.0	7.5	7.9	7.4	7.3	7.6	8.0	7.3	6.9	7.8	7.68	7.69	7.80	7.79	
Total Organic Carbon	NA	NA	NA	7995	9729	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	11074	8967	4495	9468	

Notes: ^(a) Units in mg/kg except for pH.

J = Estimated value.

NA = Not analyzed.

TABLE 4.5-11
METAL CONCENTRATIONS IN SPRING SEDIMENTS

TABLE 4.5-11

Parameter	Spring Group										Background Soil EPA
	I - Batiste System		II - Swanson Road System	III - East Side System		IV - Papoose System					
	Batiste Spring	Batiste Springs Drainage	Swanson Road Spring	Springs near STP	Spring-Fed Pond at FMC Park	Papoose Spring	Papoose Springs Drainage	Papoose Springs Discharge	Siphon Road Spring	Twenty Spring (East)	
	SD14 (sand and gravel)	SD11 (clayey sandy gravel)	SD15 (sand w/ silt)	SD13 (sand)	SD09 (loam)	SD07 (clayey sandy gravel)	*	SD05 (silty clay)	SD04 (loam)	SD02 (silty clay)	
Aluminum, total	8230	3350	2970	2630	8600	2530	NS	6720	4150	5770	13,900
Arsenic, total	5.1	ND	1.7	1.5	7.6	9.1	NS	5.1	8.2	13.8	7.7
Barium, total	324 J	55.8 J	85.3 J	77.9	134 J	52.1 J	NS	93.5 J	83.2 J	86.3 J	188
Beryllium, total	0.79	ND	0.71	0.35	1.40	0.74	NS	0.84	ND	2.20	1
Boron, total	3.7	4.1	4.0	3.4	4.6	5.6	NS	5.9	5.0	ND	12.8
Cadmium, total	0.35	ND	ND	1.5	0.37	ND	NS	ND	ND	ND	1.9
Chromium, total	14.1 J	13.5 J	9.0 J	15.2	12.9 J	11.5 J	NS	15.1 J	20.6 J	54.0 J	27.5
Cobalt, total	ND	ND	ND	2.1	ND	ND	NS	ND	ND	ND	7.6
Copper, total	13.0 J	6.8 J	8.1 J	6.8	7.8 J	5.3 J	NS	9.3 J	7.5 J	11.8 J	12.6
Iron, total	6850 J	8524 J	6620 J	5530	9710 J	7760 J	NS	8220 J	5920 J	10400 J	
Lead, total	29.5 J	5.9 J	12.8 J	24.7 J	8.7 J	7.6 J	NS	50.5 J	7.3 J	ND	29.1
Lithium, total	9.1	5.1	3.9	3.9	9.9	4.0	NS	8.4	6.9	4.8	16.1
Manganese, total	117 J	75.1 J	405 J	116 J	281 J	52 J	NS	100 J	56 J	22.7 J	482
Mercury, total	ND	ND	ND	ND	ND	ND	NS	ND	ND	ND	0.16
Molybdenum, total	ND	ND	ND	ND	ND	ND	NS	ND	ND	ND	2.15
Nickel, total	7.6	5.2	4.1	2.5	6.6	5.1	NS	6.0	ND	ND	15.5
Selenium, total	ND	ND	ND	3.50	ND	ND	NS	ND	ND	ND	1.36
Silver, total	0.20	0.60	2.1	ND	1.1	0.17	NS	0.75	0.47	ND	1.9
Thallium, total	R	R	0.14 J	ND	0.14	R	NS	0.30 J	R	R	0.27
Vanadium, total	20.7	21.6	20.5	28.0	24.3	17.4	NS	26.2	25.2	192	45.4
Zinc, total	18.5 J	107.0 J	15.4 J	23.5	27.8 J	25.1 J	NS	54.3 J	31.4 J	37.4 J	52.8

Notes: Units in mg/kg.
 J = Estimated Value
 ND = Not detected/below detection level.
 NS = Not sampled.
 R = Rejected value.
 * = Not assigned.

TABLE 4.5-12
CONCENTRATIONS OF NUTRIENTS AND OTHER PARAMETERS IN SPRING SEDIMENTS

Parameter	SPRING GROUP									
	I - Batiste System		II - Swanson Road System	III - East Side System		IV - Papoose System				
	Batiste Spring	Batiste Springs Drainage	Swanson Road Spring	Springs near STP	Spring-Fed Pond at FMC Park	Papoose Spring	Papoose Springs Drainage	Papoose Springs Discharge	Siphon Road Spring	Twenty Springs (East)
	SD14 (sand and gravel)	SD11 (clayey sandy gravel)	SD15 (sand w/ silt)	SD13 (sand)	SD09 (loam)	SD07 (clayey sandy gravel)	*	SD05 (silty clay)	SD04 (loam)	SD02 (silty clay)
Orthophosphate (PO ₄ as P), (mg/kg)	3.4	ND	4.9	1.2	2.3	0.5	NS	0.9	ND	ND
Total phosphorus, (mg/kg)	286	537	955	3950	393	469	NS	577	387	64.5
Fluoride, (mg/kg)	89	155	333	800	222	206	NS	265	121	75.3
pH	7.2	7.8	8.0	7.9	8.2	7.4	NS	7.4	7.5	8.5

Notes: Calcium, magnesium, and total organic carbon were not analyzed.

ND = Not detected/ below detection limit.

NS = Not sampled.

* = Not assigned.

Table 4.5-13
Summary of t-tests for Sediment Groups

Table 4.5-13

Parameter	Upstream Sediments vs. Spring Sediments Calculated T-statistic	Hypothesis Supported At 95% Confidence Interval Critical One-tailed T-value = 1.76	Near Site Sediments vs. Upstream Sediments Calculated T-statistic	Hypothesis Supported At 95% Confidence Interval Critical One-tailed T-value = 1.83	Upstream Sediments vs. Downstream Sediments Calculated T-statistic	Hypothesis Supported At 95% Confidence Interval Critical One-tailed T-value = 1.76
Aluminum	2.42	Upstream Sediments Higher	-1.26	No difference	-1.26	No difference
Arsenic	-0.95	No difference	-0.44	No difference	1.02	No difference
Barium	0.45	No difference	0.09	No difference	-1.12	No difference
Beryllium	-2.06	Spring Sediments Higher	-0.07	No difference	1.58	No difference
Boron	1.04	No difference	-1.45	No difference	0.55	No difference
Chromium	-0.61	No difference	0.06	No difference	-0.23	No difference
Copper	3.36	Upstream Sediments Higher	0.77	No difference	-0.73	No difference
Fluoride	1.42	No difference	-1.09	No difference	-0.74	No difference
Lead	1.21	No difference	0.24	No difference	-1.77	No difference
Lithium	2.79	Upstream Sediments Higher	-1.55	No difference	-0.92	No difference
Manganese	3.25	Upstream Sediments Higher	-1.62	No difference	-2.83	Upstream Sediments Higher
Nickel	1.83	Upstream Sediments Higher	-1.57	No difference	-0.64	No difference
Orthophosphate	0.75	No difference	-1.72	No difference	0.75	No difference
Silver	1.61	No difference	1.33	No difference	-0.79	No difference
Total Phosphorus	-1.08	No difference	1.12	No difference	1.34	No difference
Vanadium	-0.75	No difference	-0.06	No difference	-2.44	Upstream Sediments Higher
Zinc	0.36	No difference	-0.46	No difference	-0.09	No difference

Non-parametric Test	Kruskal-Wallis Statistic	Hypothesis Supported at 95% Confident Level Critical value 7.81
Selenium	6.77	No groups are different
Mercury	5.53	No groups are different
Thallium	6.09	No groups are different
Cadmium	0.78	No groups are different

Notes:

Upstream Sediment Samples = SD21, SD22, SD23, SD24, SD25, SDA1, SDA2

Spring Sediment Samples = SD2, SD4, SD5, SD6, SD7, SD9, SD11, SD13, SD14, SD15

Downstream Sediments Samples = SD1, SD3, SD8, SD10, SD12 SDC1, SDC2, SDC4, SDB1

Near Site Sediment Samples = SD16, SD18, SD19, SD20 (SD17 known to be impacted and not included in analysis)

Antimony was not detected in any sediment sample, no statistical tests performed.

Molybdenum was detected in one upstream sediment sample, no statistical tests performed

T-tests were performed assuming normal distribution of population and U-flagged data were not treated; (i.e., U'd data were used at the full reported detection limit).

Section 4 Nature and Extent of Constituents of Potential Concern

TABLE 4.5-14
ANALYTICAL RESULTS FOR PHASE II SEDIMENT SAMPLES
TAKEN AT FMC OUTFALL (mg/kg)

Parameter (mg/kg)	17A	17B	17C
Aluminum, total	10700	5990	4710
Antimony, total	7.4 U	6.2 U	7 U
Arsenic, total	8.6	2.7	3.9
Barium, total	801	99.8	132
Beryllium, total	1.2	0.28	0.29
Boron, total	25.2	4.2	3.8
Cadmium, total	104	0.91 U	0.46 U
Calcium	131000	102000	208000
Chromium, total	112	10.4	9.4
Cobalt, total	25.2	4.1	4.4
Copper, total	352	6.1	4.1
Fluoride	7760	310	230
Iron, total	11800	7070	6600
Lead, total	21.7 J	8.7J	1.2 J
Lithium, total	10.4	5.7	4.8
Magnesium	4700	4550	4850
Manganese, total	21500	219	437
Mercury, total	0.19	0.04 U	0.17
Molybdenum, total	2.4 U	1.8 U	2 U
Nickel, total	62.3	7.2 U	6.9 U
Orthophosphate	12.6	2.15	0.76
Total Phosphorus	15900	490	190
Potassium	2310	1320	1030
Selenium, total	2	0.72 U	0.76 U
Silver, total	4 U	0.62 U	1.7 U
Sulfate	20 U	20 U	20 U
Thallium, total	0.82 J	0.14 J	0.3 J
Vanadium, total	111	4.8	0.31 U
Zinc, total	1800 J	40.7 J	31.9 J

Notes: J = Estimated value.
U = Undetected.

TABLE 4.5-15
RADIOLOGICAL ACTIVITIES IN RIVER SEDIMENTS (pCi/g)

Parameter	Upstream River Stations											River Station SW17
	SD25 (clayey sand)	SD24 (clay)	SD23 (clay)	SDA2 (silty clay)	SDA1 (silty clay)	SD22 (silty clay)	SD21 (silt and sand)	SD20 (sand w/ silt and gravel)	SD19 (silty clay)	SD18 (sand w/ gravel)	SD16 (silty clay)	SD17 (sandy clay)
Gross alpha	6.52±0.90	12.5±1.3	13.6±1.3	6.33±2.96	ND	6.52±0.98	8.44±1.17	6.07±0.83	12.0±2.3	8.94±2.22	8.95±2.03	29.2±3.6
Gross beta	10.8±1.2	25.3±1.5	25.1±1.5	13.4±2.8	10.2±2.6	12.5±1.1	13.3±1.1	9.24±1.00	16.9±2.4	14.1±2.6	14.9±2.4	30.0±3.2
Cesium-137	ND	0.139±0.041	0.744±0.080	NA	NA	ND	ND	ND	0.144±0.046	ND	0.111±0.056	ND
Europium-155	ND	ND	ND	NA	NA	ND	ND	ND	ND	0.212±0.080	ND	ND
Potassium-40	9.96±0.89	16.7±1.1	18.4±1.3	11.7±1.8	9.65±1.57	8.08±0.67	14.4±1.1	9.6±0.7	13.8±1.0	13.2±1.0	14.1±1.1	10.3±0.8

Parameter	Downstream River Stations								
	SD12 (sand)	SD10 (fine sand)	SD08 (loam)	SD03 (loam, sand, and gravel)	SD01 (loam)	SDB1 (silty clay)	SDC1 (silt w/ fine sands)	SDC2 (silt w/ fine sands)	SDC4 (silt w/ fine sands)
Gross alpha	9.94±1.89	9.63±2.25	9.40±1.66	10.7±1.9	8.73±2.42	8.15±3.22	ND	6.03±3.57	ND
Gross beta	15.3±1.9	12.7±2.2	14.4±1.9	16.1±2.2	14.3±2.4	13.3±2.8	ND	13.5±2.9	7.3±2.4
Cesium-137	ND	ND	ND	ND	0.103±0.032	NA	NA	NA	NA
Europium-155	ND	ND	ND	ND	ND	NA	NA	NA	NA
Potassium-40	11.2±0.9	8.92±0.71	10.8±0.8	11.4±0.8	12.8±0.9	9.58±1.61	2.64±1.27	5.87±1.05	5.79±1.32

Notes: Units in pCi/g.

NA = Not analyzed.

ND = Not detected/below detection level.

NS = Not sampled.

TABLE 4.5-16
RADIOLOGICAL ACTIVITIES IN SPRING SEDIMENTS (pCi/g)

Parameter (pCi/g)	SPRING GROUP									
	I - Batiste System		II - Swanson Road System	III - East Side System		IV - Papoose System				
	Batiste Spring	Batiste Springs Drainage	Swanson Road Spring	Spring near STP	Spring-Fed Pond at FMC Park	Papoose Spring	Papoose Springs Drainage	Papoose Springs Discharge	Siphon Road Spring	Twenty Springs (East)
	SD14 (sand and gravel)	SD11 (clayey sandy gravel)	SD15 (sand w/ silt)	SD13 (sand)	SD09 (loam)	SD07 (clayey sandy gravel)	*	SD05 (silty clay)	SD04 (loam)	SD02 (silty clay)
Gross alpha	19.8±2.5	15.2±3.1	11.6±2.0	14.8±1.4	11.1±1.8	15.4±2.1	NS	13.1±2.1	10.0±1.9	18.3±2.5
Gross beta	18.4±1.9	18.9±2.8	16.1±1.9	18.3±1.3	19.5±2.1	14.7±2.1	NS	18.2±2.3	15.9±1.9	19.7±2.1
Cesium-137	ND	ND	ND	ND	ND	0.336±0.065	NS	0.377±0.055	0.111±0.040	ND
Europium-155	ND	ND	ND	ND	0.306±0.077	ND	NS	0.49±0.12	ND	ND
Potassium-40	12.5±1.0	9.77±0.75	13.4±1.0	9.96±0.79	15.1±1.1	9.3±1.1	NS	13.6±1.1	12.0±1.0	13.6±1.2

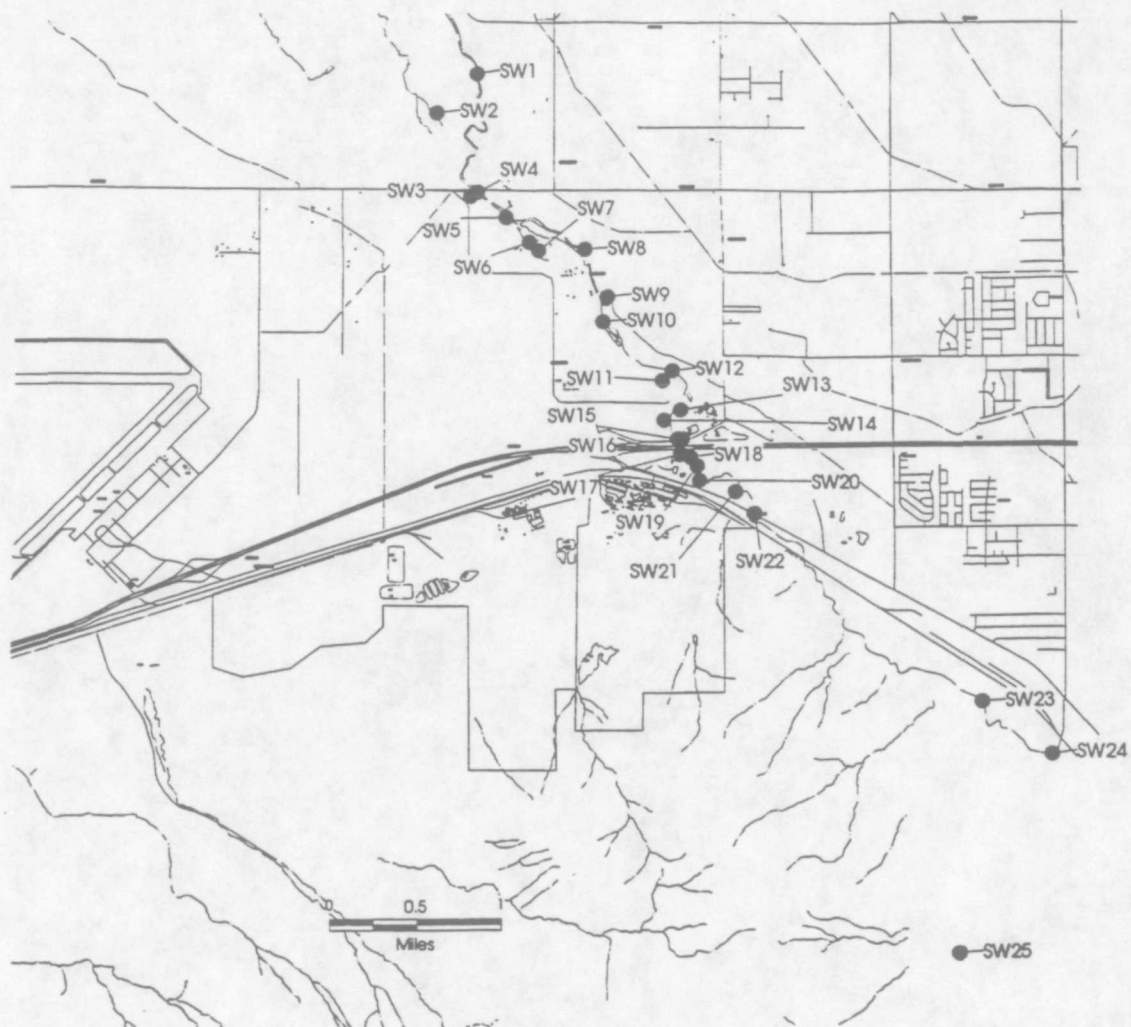
Notes: NA = Not analyzed.

ND = Not detected/below detection level.

NS = Not sampled.

* = Not assigned.

Surface and Subsurface Characterizations
Figures for Section 4.5



Note: For larger scale and more detail,
please refer to Figure 2.4-1

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EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Surface Water
Sampling Locations



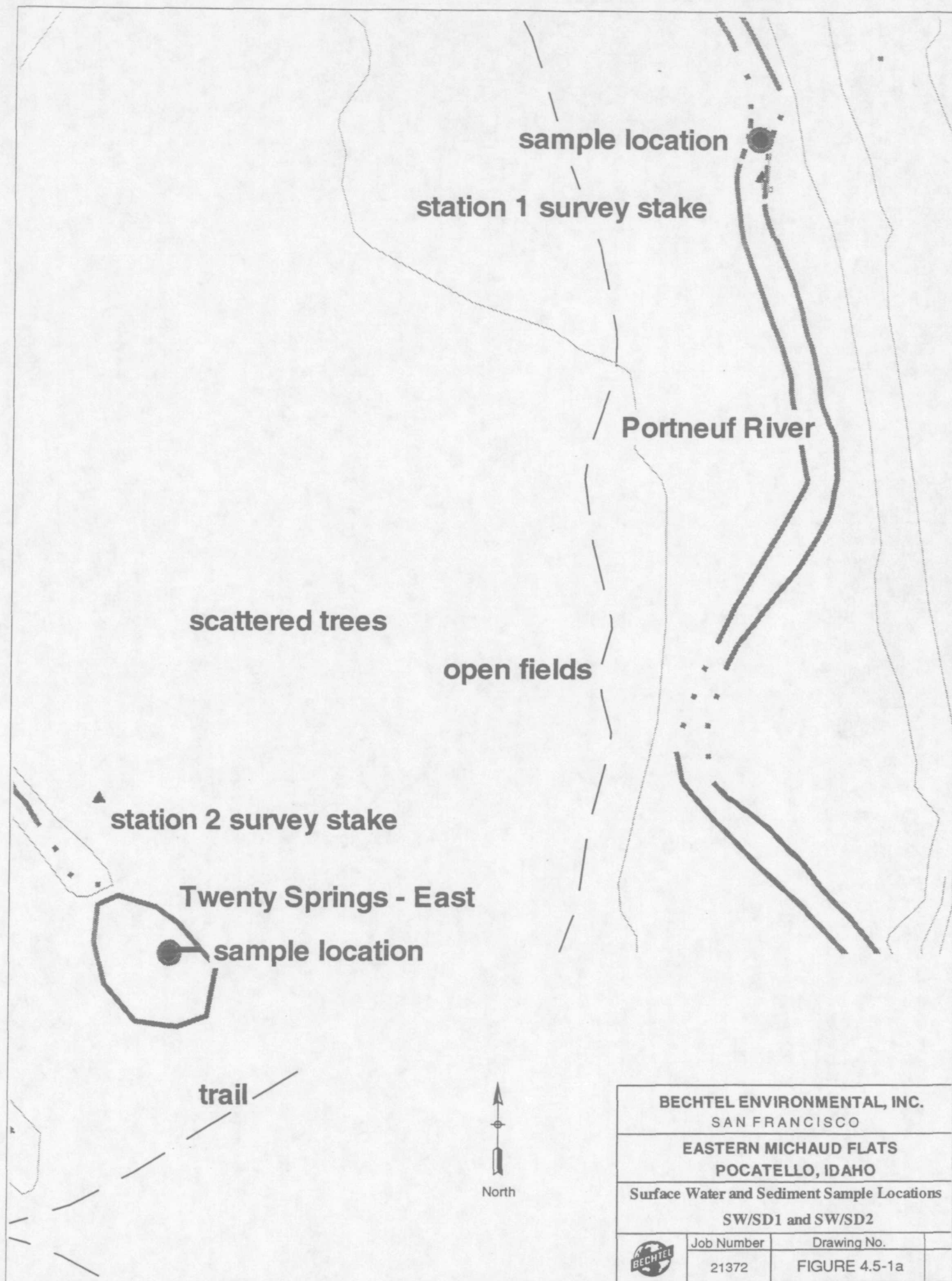
JOB No.


21372

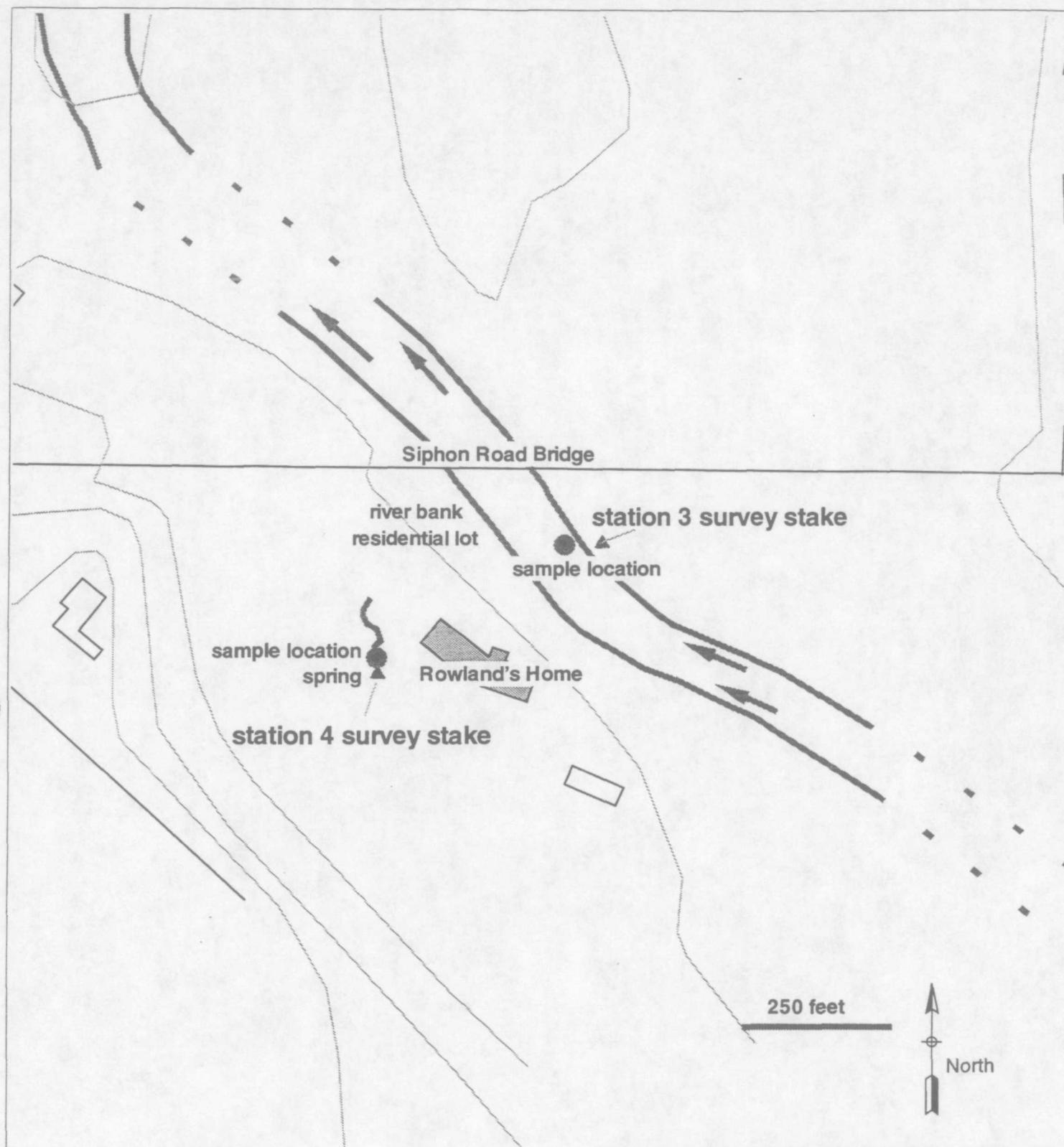
DRAWING NO.

FIGURE 4.5-1

REV.



BECHTEL ENVIRONMENTAL, INC. SAN FRANCISCO		
EASTERN MICHAUD FLATS POCATELLO, IDAHO		
Surface Water and Sediment Sample Locations SW/SD1 and SW/SD2		
	Job Number	Drawing No.
	21372	FIGURE 4.5-1a



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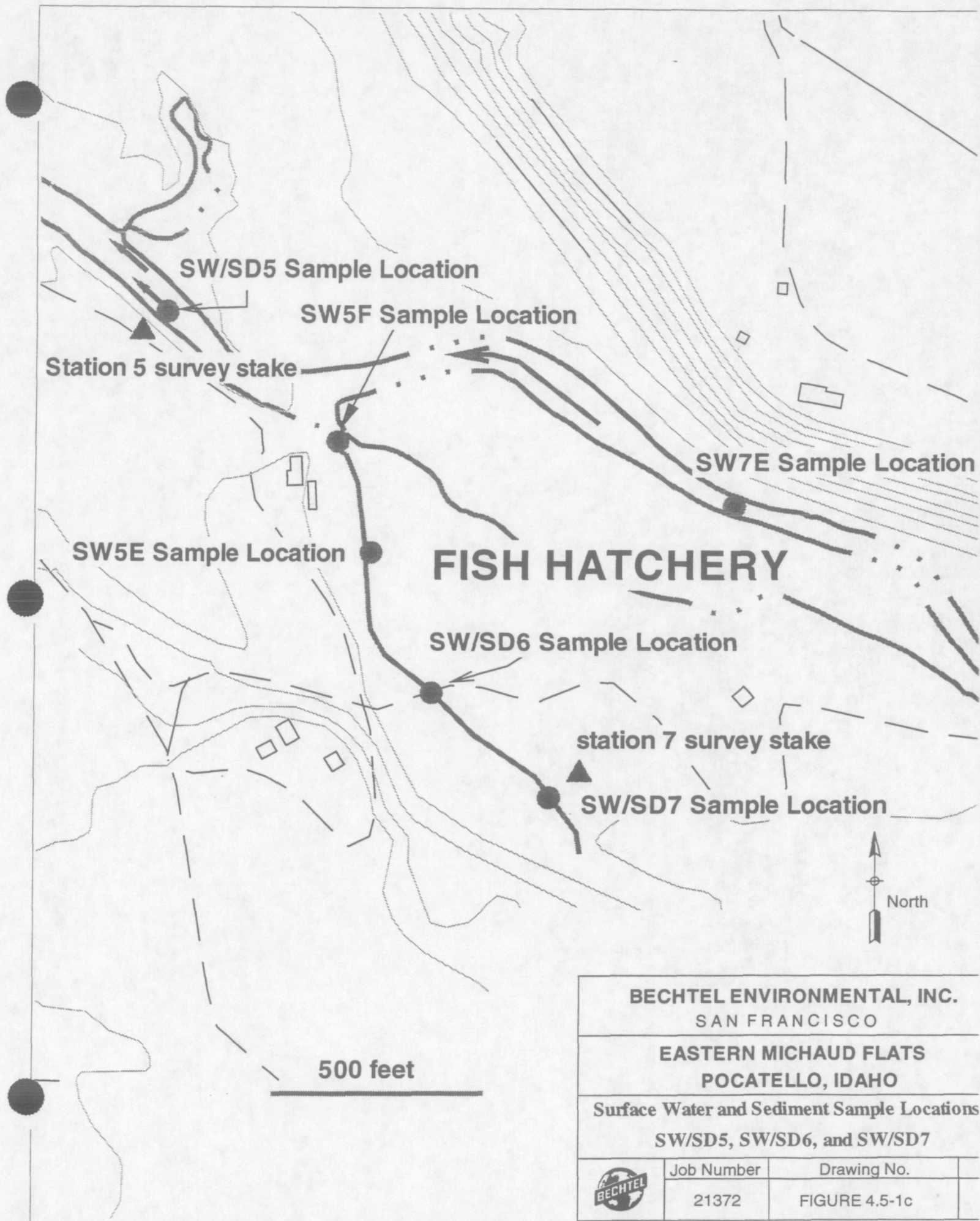
EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Surface Water and Sediment Sample Locations
SW/SD3 and SW/SD4



Job Number
21372

Drawing No.
FIGURE 4.5-1b



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EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Surface Water and Sediment Sample Locations

SW/SD5, SW/SD6, and SW/SD7

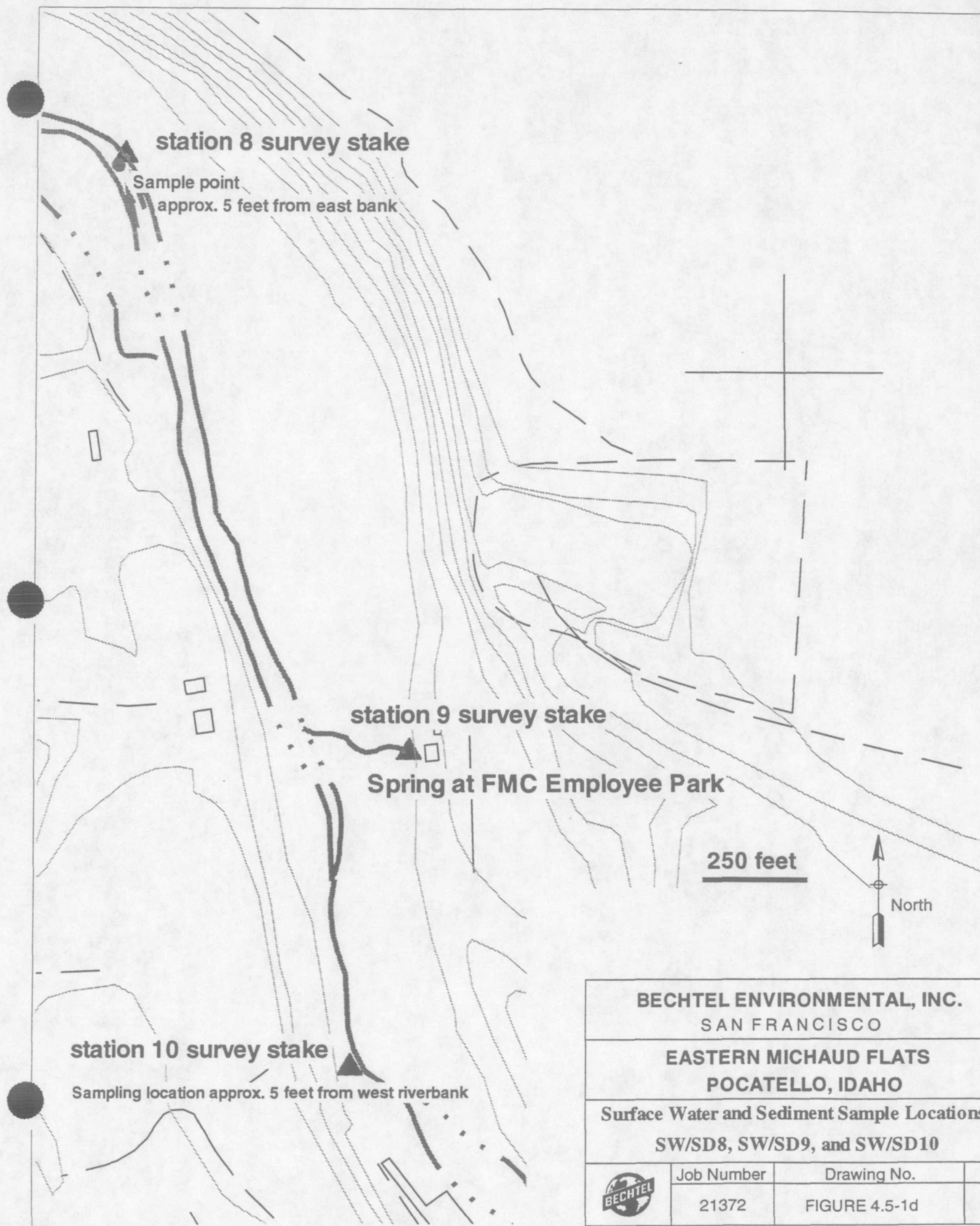


Job Number

Drawing No.

21372

FIGURE 4.5-1c



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EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Surface Water and Sediment Sample Locations
SW/SD8, SW/SD9, and SW/SD10

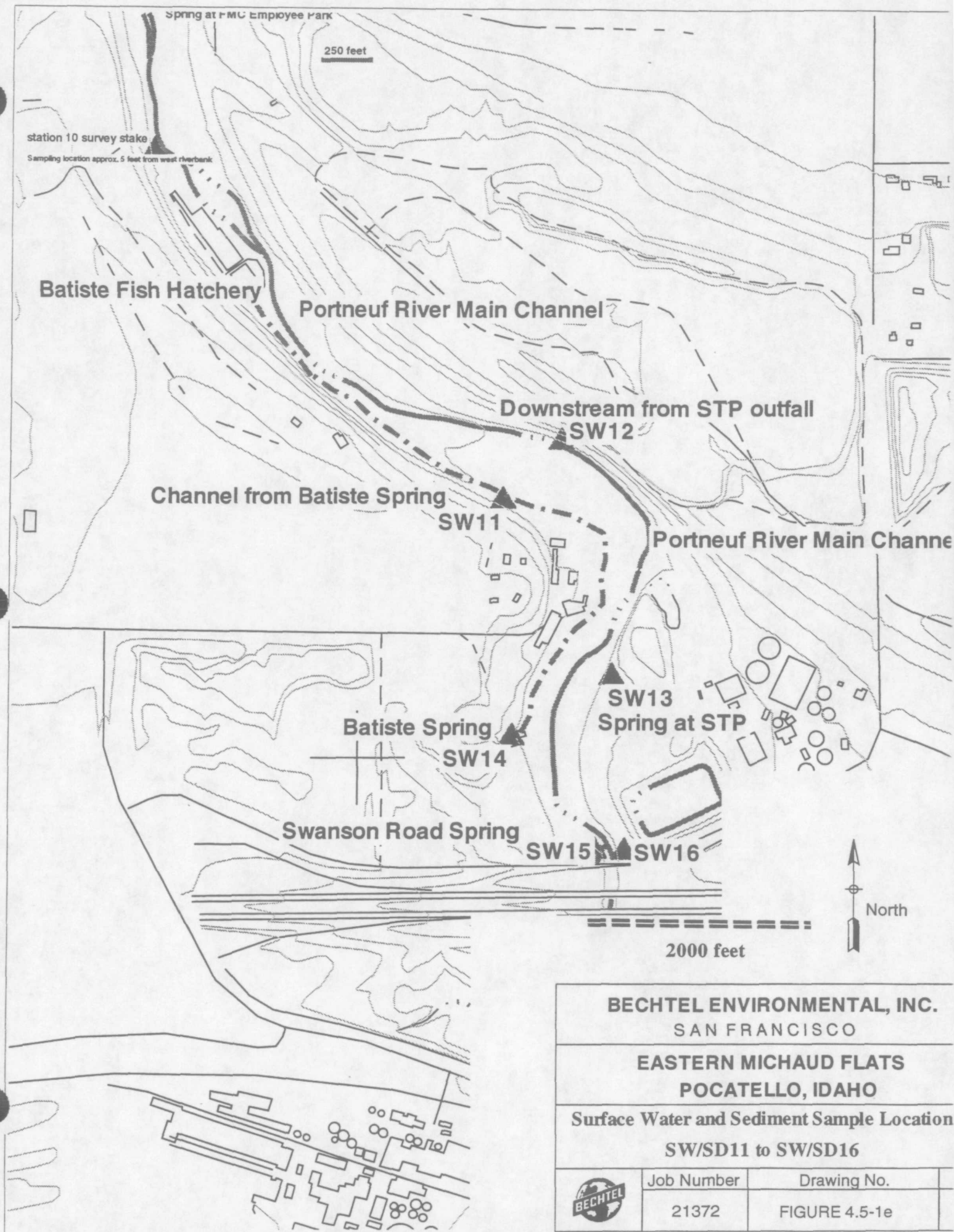


Job Number

21372

Drawing No.

FIGURE 4.5-1d



I-86

station 17 survey stake

IWW Outfall

▲ station 18 survey stake

Portneuf River

station 19 survey stake

sample location

station 20 survey stake

North

Former East
Overflow Pond

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EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Surface Water and Sediment Sample Locations
SW/SD17, SW/SD18, SW/SD19, and SW/SD20

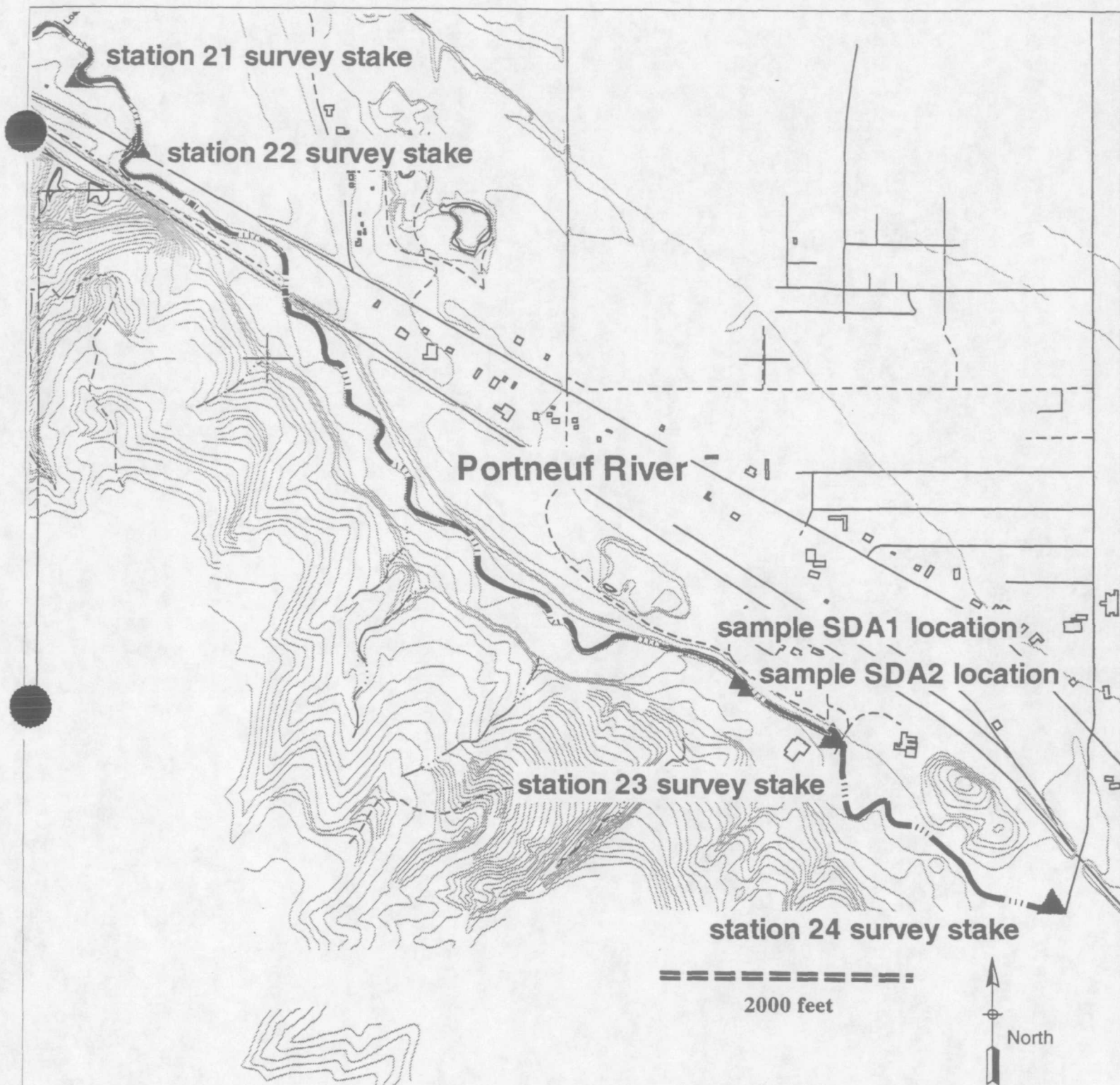


Job Number

Drawing No.

21372

FIGURE 4.5-1f



BECHTEL ENVIRONMENTAL, INC.
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EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Surface Water and Sediment Sample Locations
SW/SD21, SW/SD22, SW/SD23, SW/SD24



Job Number

21372

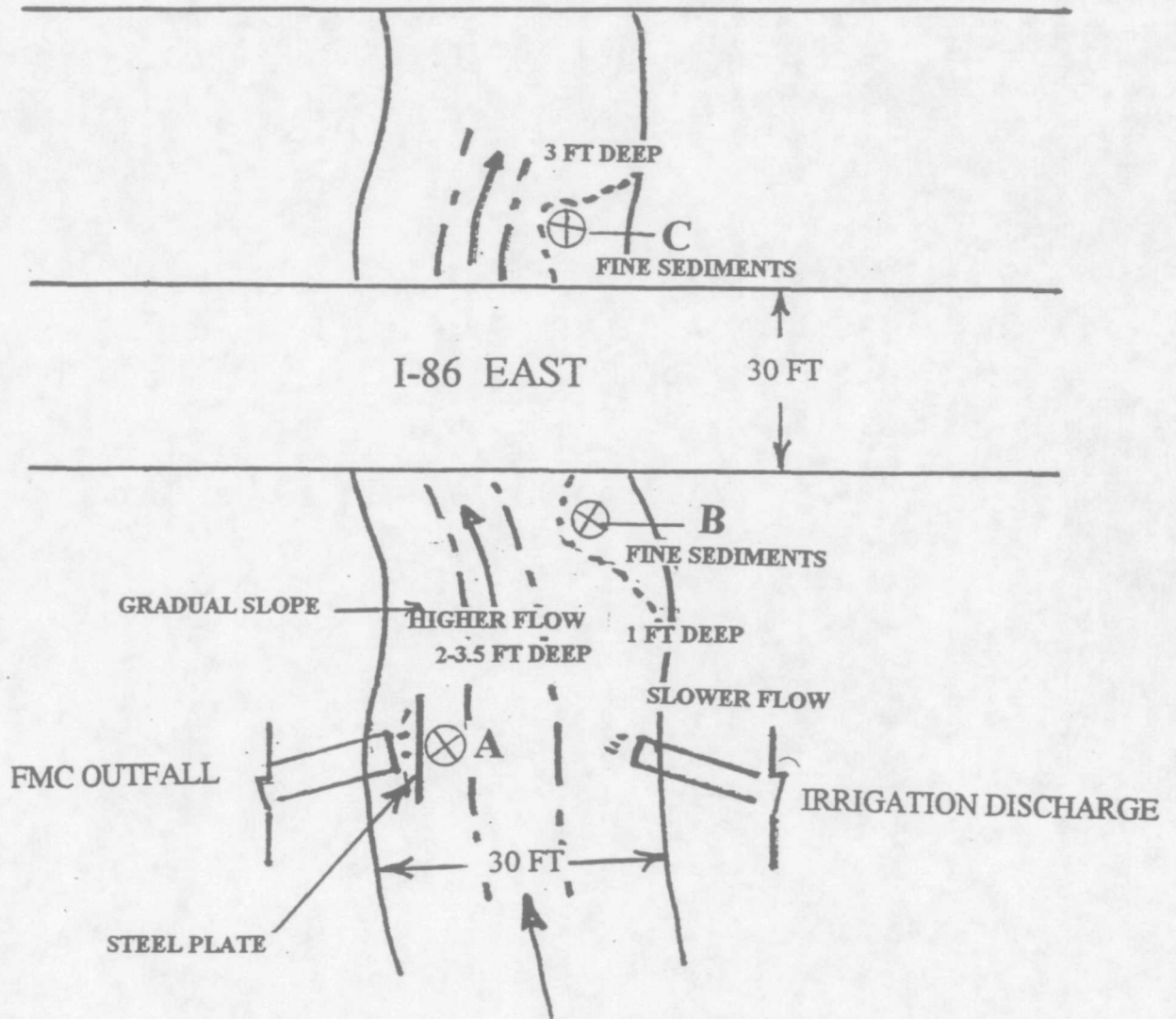
Drawing No.

FIGURE 4.5-1g

PORTNEUF RIVER

I-86 WEST

I-86 EAST



NOT TO SCALE



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EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Locations of Phase II FMC
Outfall Samples



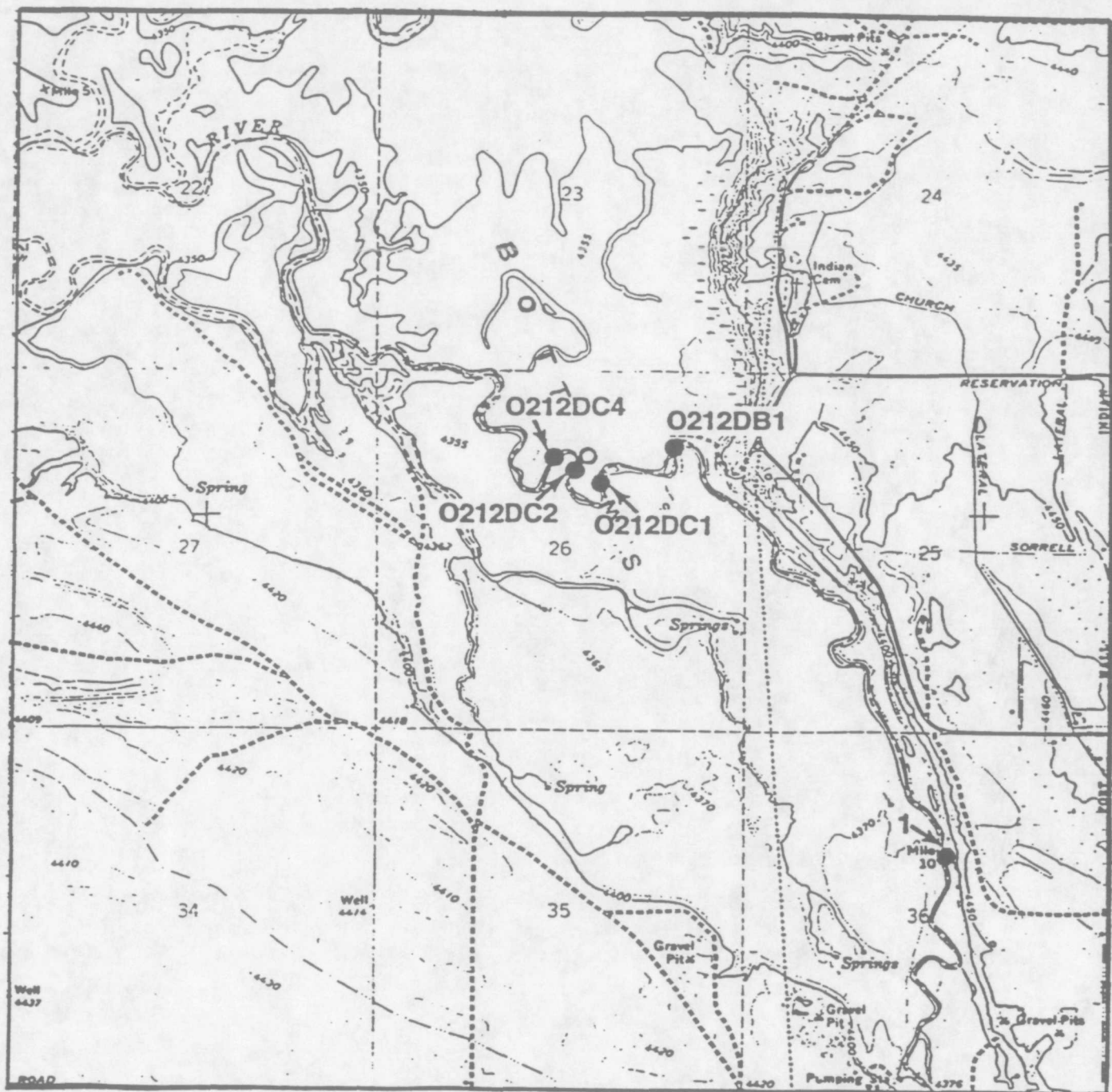
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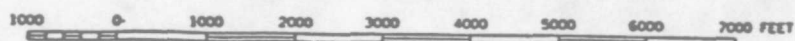
REV.

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FIGURE 4.5-1h



SCALE 1:24 000



CONTOUR INTERVAL 10 FEET



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EASTERN MICHAUD FLATS
POCATELLO, IDAHO

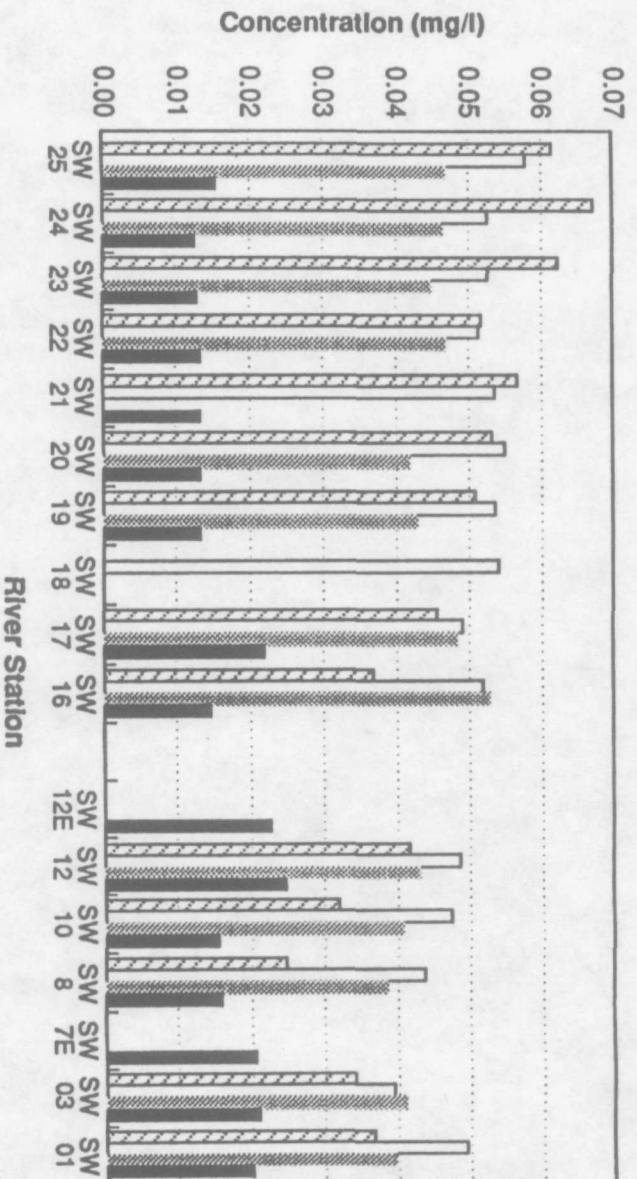
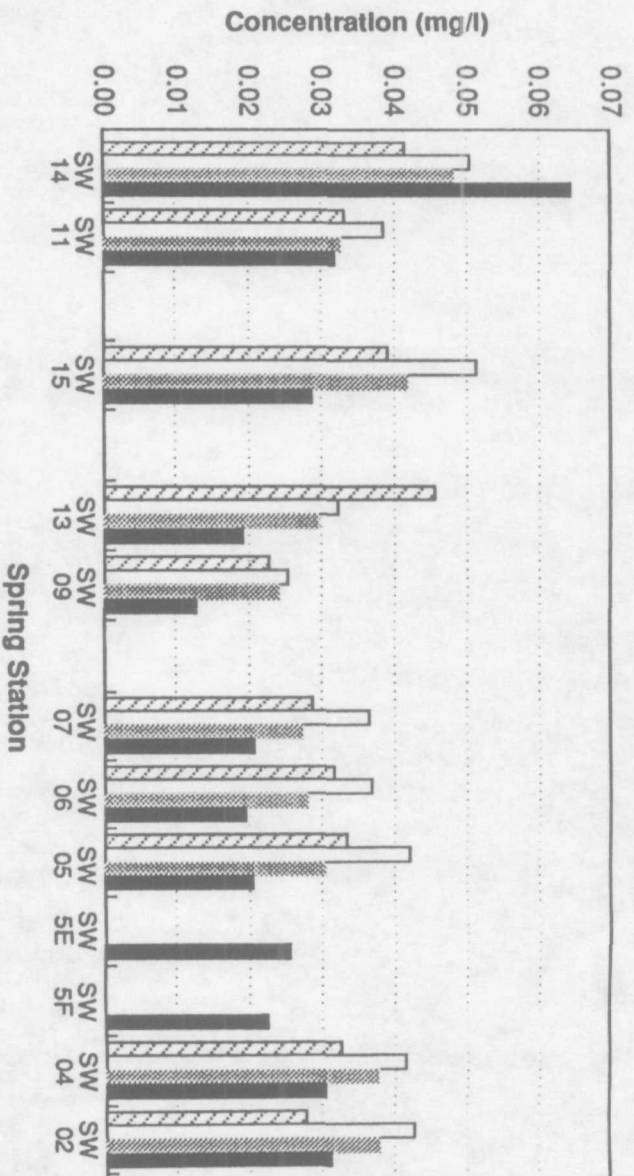
Sediment Samples Taken in the
Fort Hall Bottoms



JOB No.
21372

DRAWING NO.
FIGURE 4.5-1i

REV.



Jul-92
 Oct-92
 Feb-93
 Apr-93

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EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Lithium Concentrations in Surface
Water at Springs and River Stations



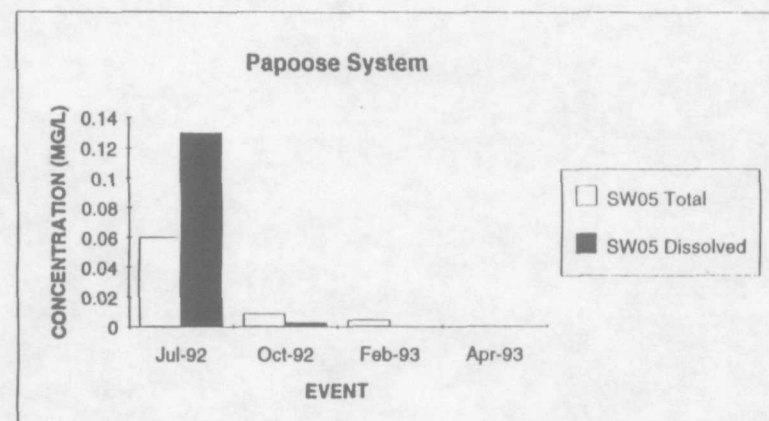
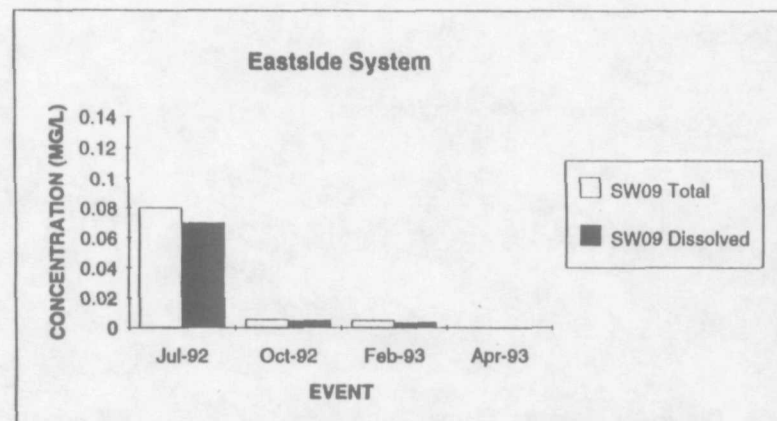
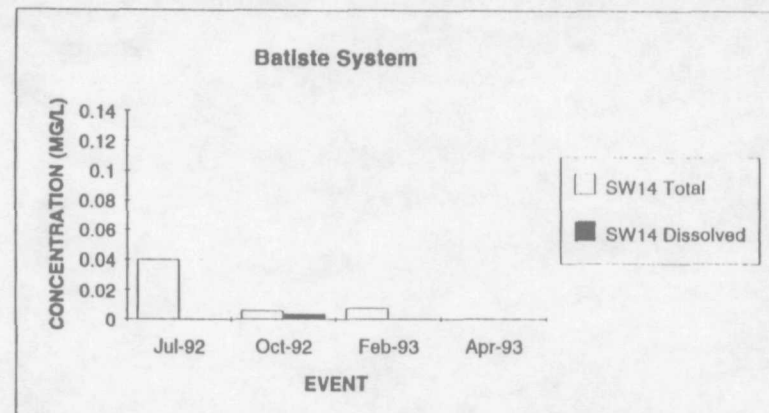
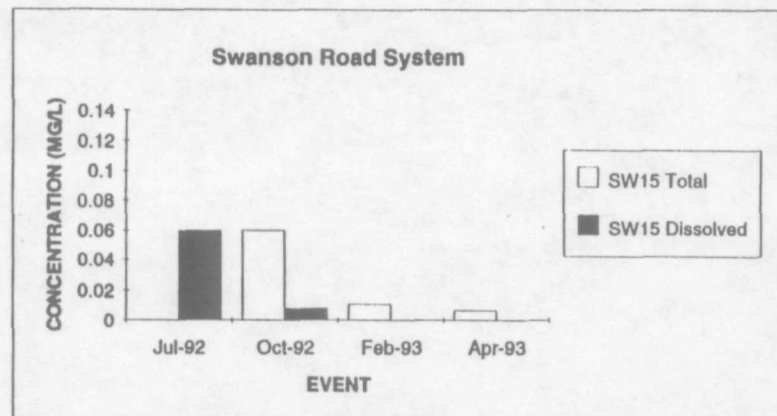
21372

FIGURE 4.5-3


JOB NO.

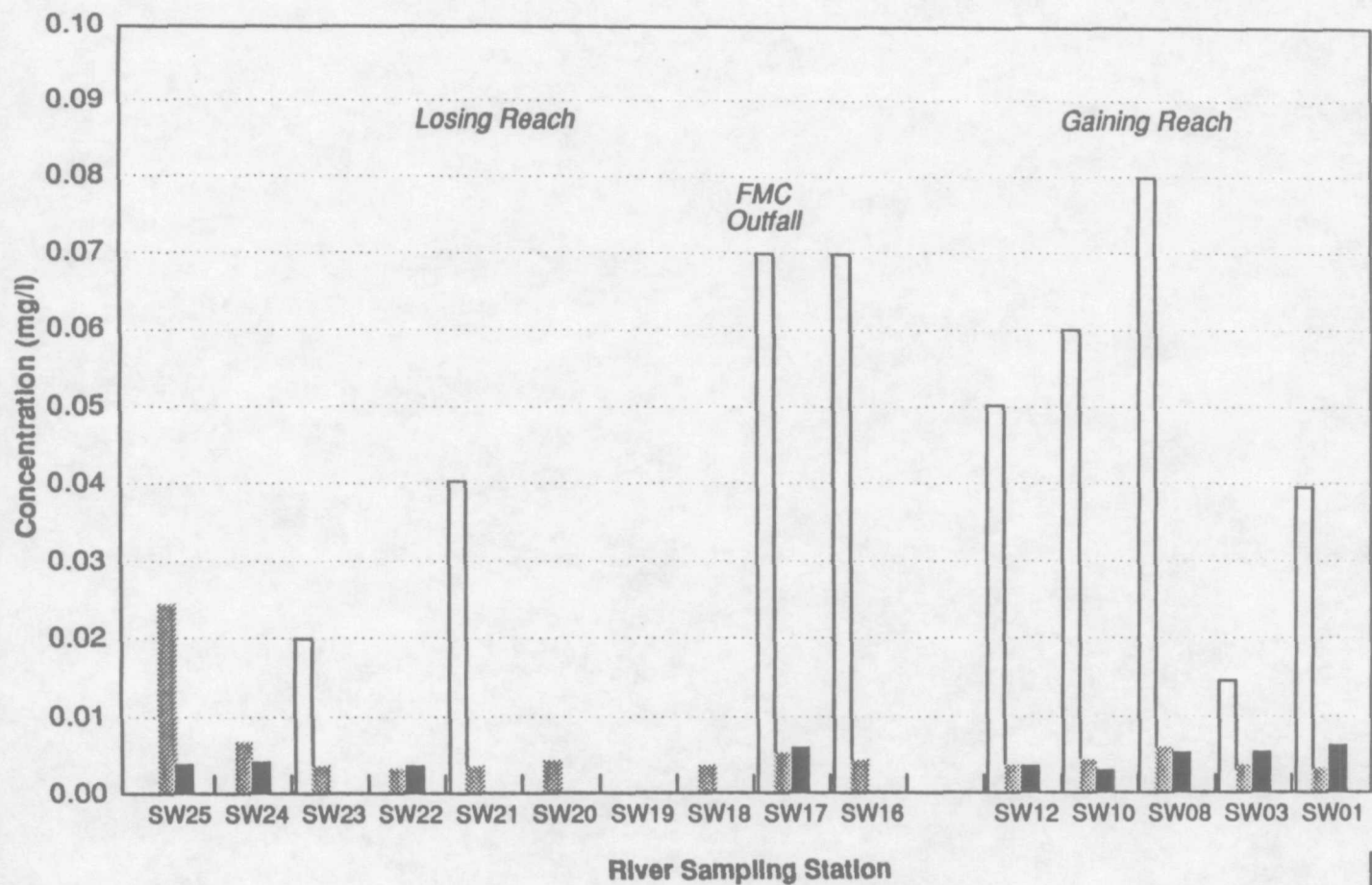
DRAWING NO.

REV.



Note: Concentrations are not denoted in this figure for samples in which results were reported as not detected.

BECHTEL ENVIRONMENTAL, INC. SAN FRANCISCO			
EASTERN MICHAUD FLATS POCATELLO, IDAHO			
Vanadium Concentrations for Each Spring System			
	JOB No.	DRAWING NO.	REV.
	21372	FIGURE 4.5-4	0



Direction of Stream Flow →

- Jul-92 Sampling Event
- ▨ Oct-92 Sampling Event
- Feb-93 Sampling Event
- Apr-93 Sampling Event (non-detect)

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EASTERN MICHAUD FLATS
POCATELLO, IDAHO

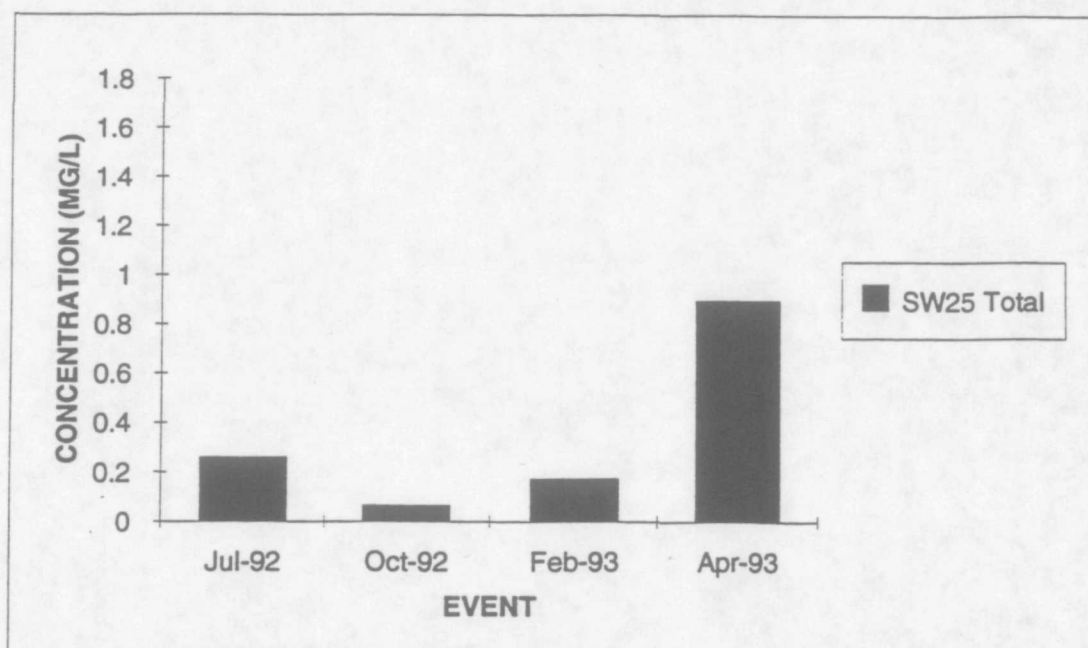
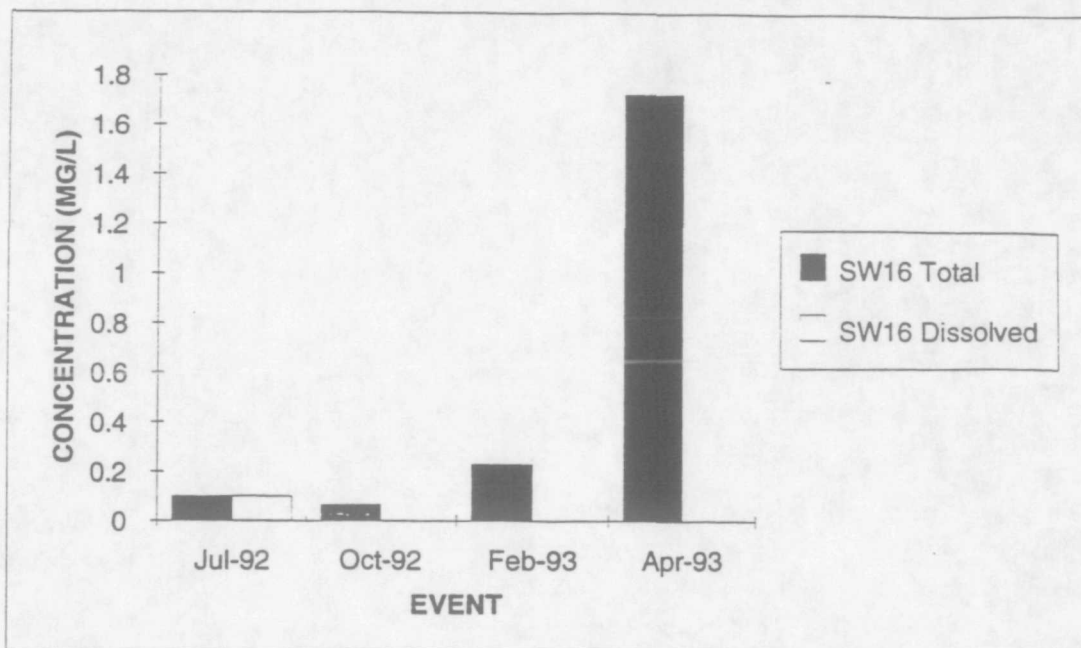
Vanadium Concentrations in Surface Water
Samples from River Sampling Stations



JOB No.
21372

DRAWING NO.
FIGURE 4.5-5

REV.



Note:
Concentrations are not denoted in this figure
for samples in which results were reported
as not detected.

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EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Aluminum Concentrations for
River Stations SW25 and SW16



JOB No.

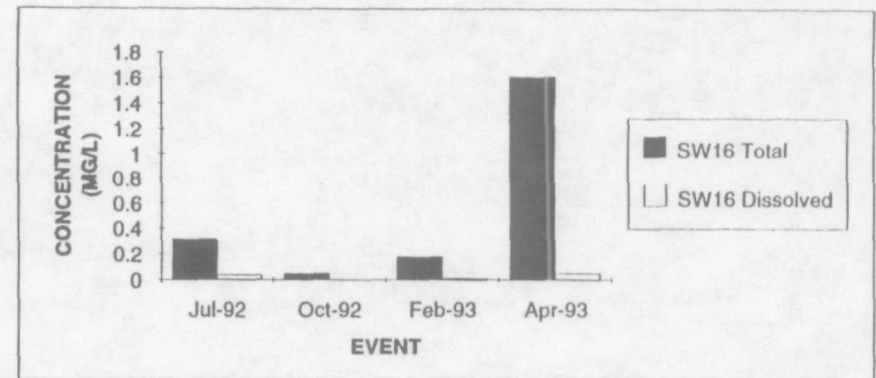
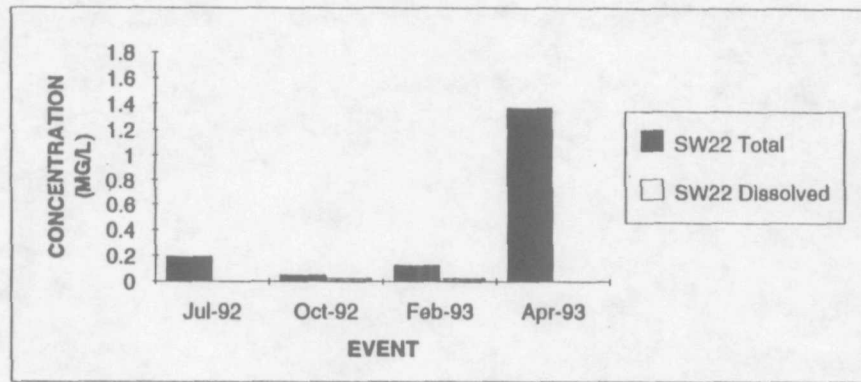
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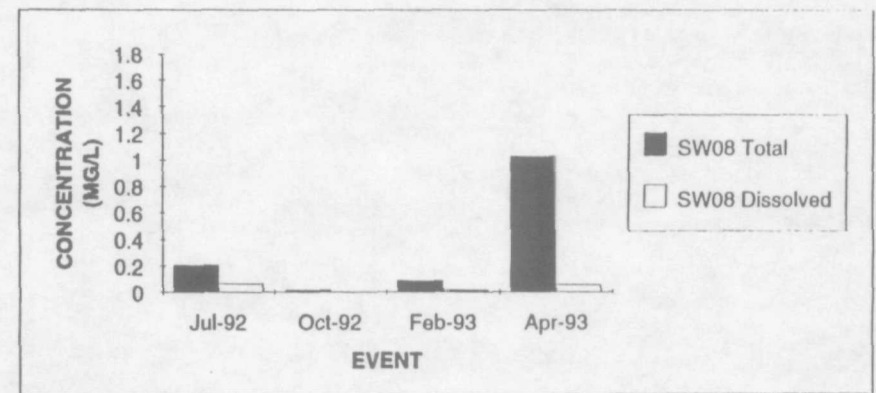
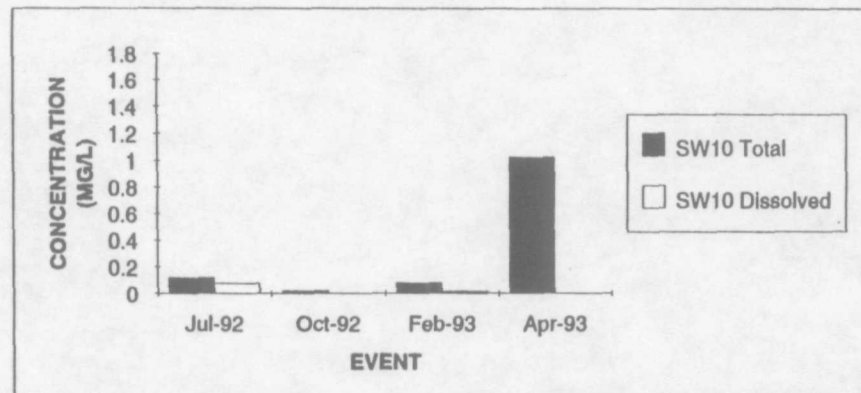
FIGURE 4.5-6

REV.

Lossing Reach



Gaining Reach



Note: Concentrations are not denoted in this figure for samples in which results were reported as not detected.

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EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Iron Concentrations in River Samples



JOB No.

21372

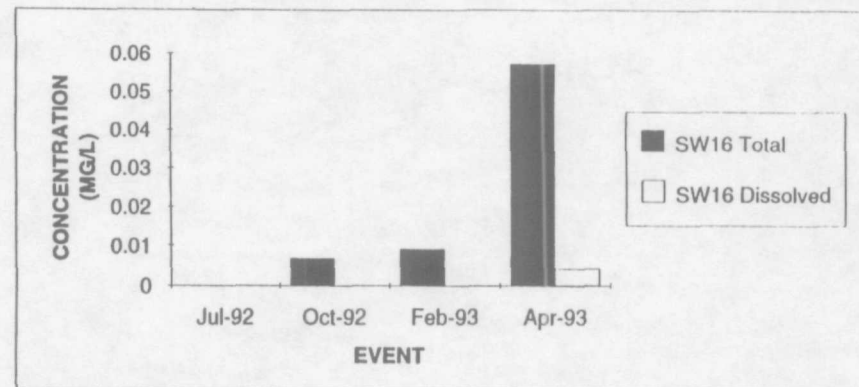
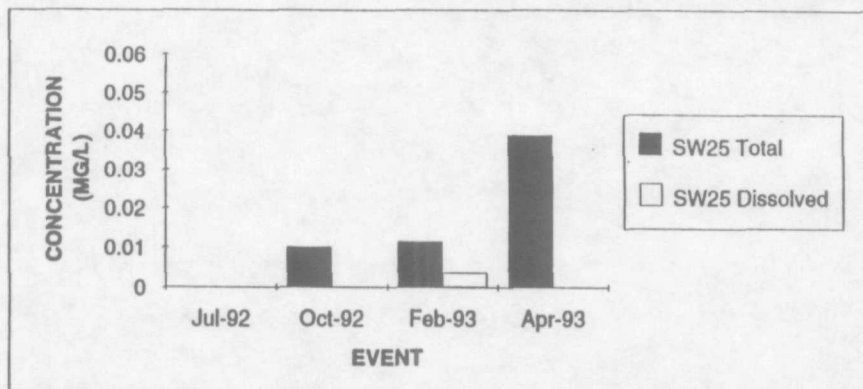
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FIGURE 4.5-7

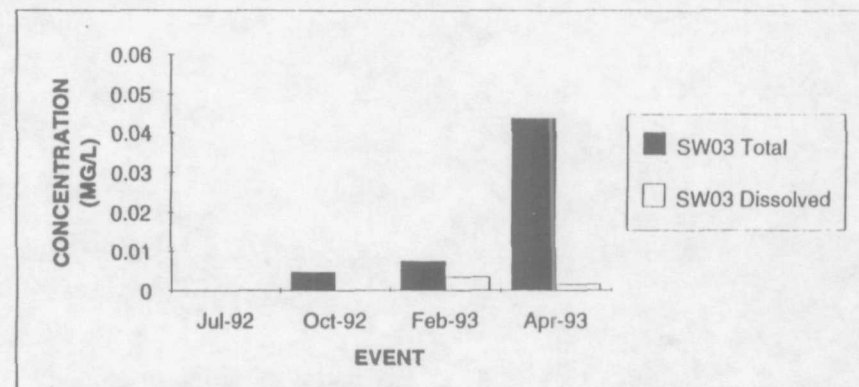
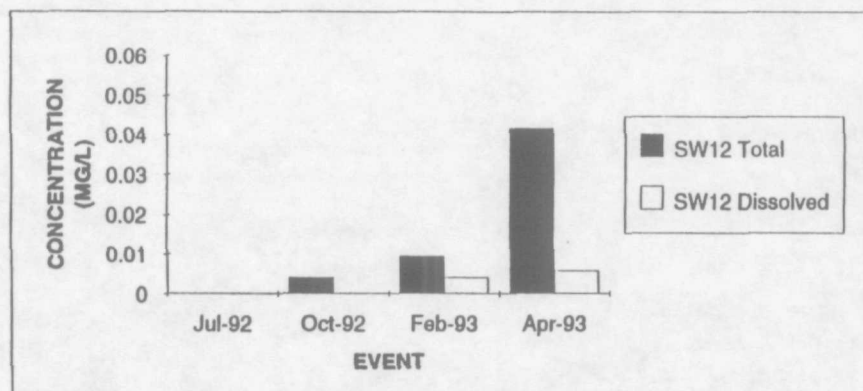
REV

0

Losing Reach



Gaining Reach



BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Manganese Concentrations in
River Samples



JOB No.

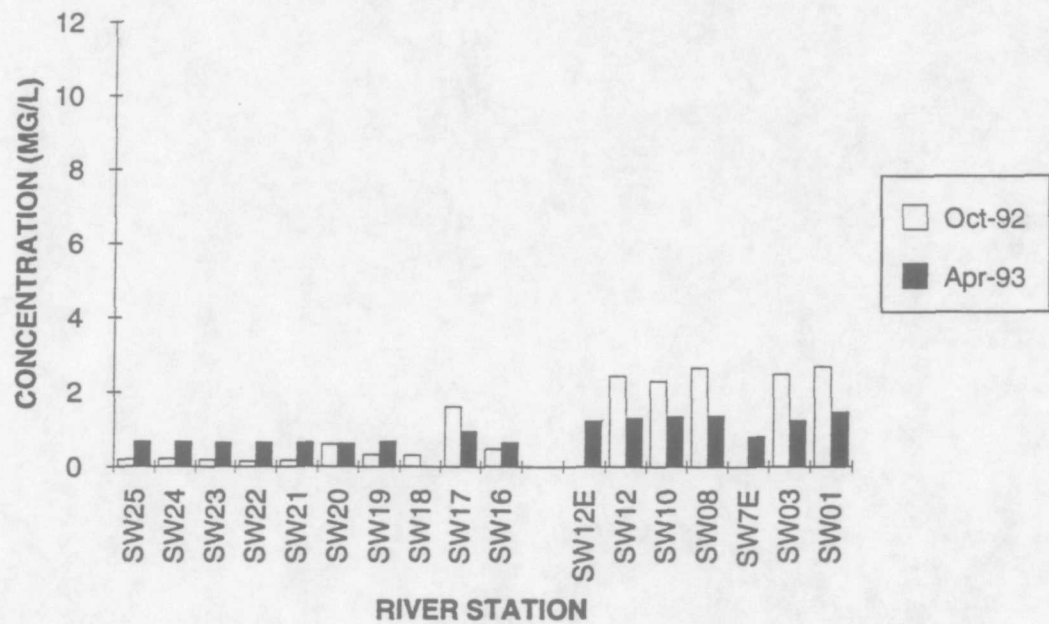
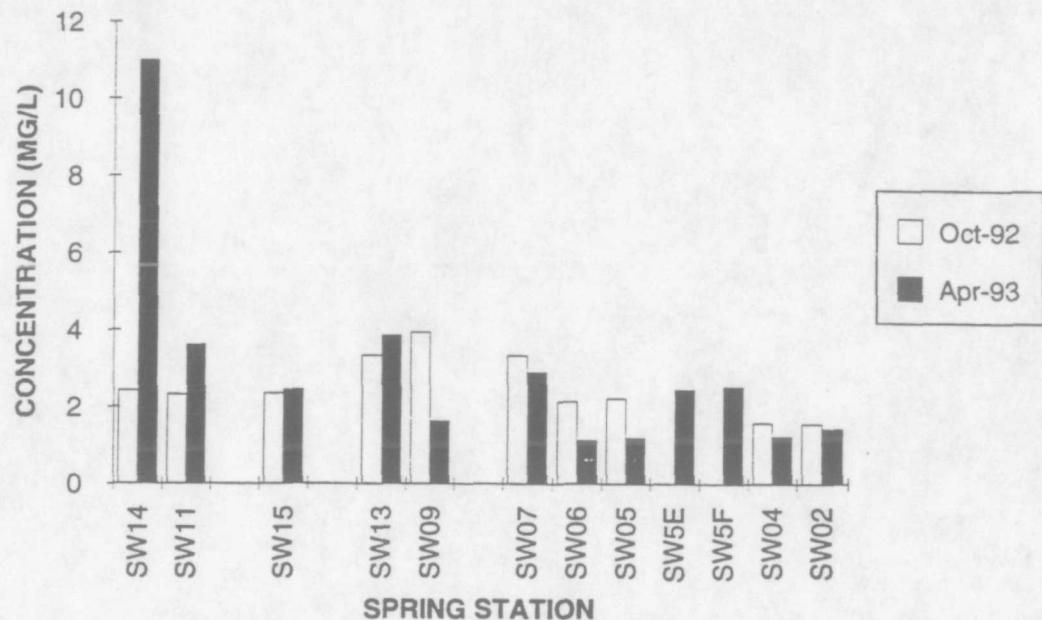
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FIGURE 4.5-8,

REV.

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BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Nitrate in Springs and River



JOB No.

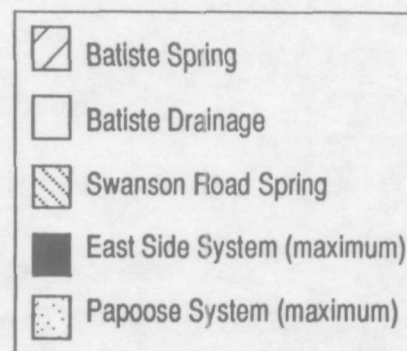
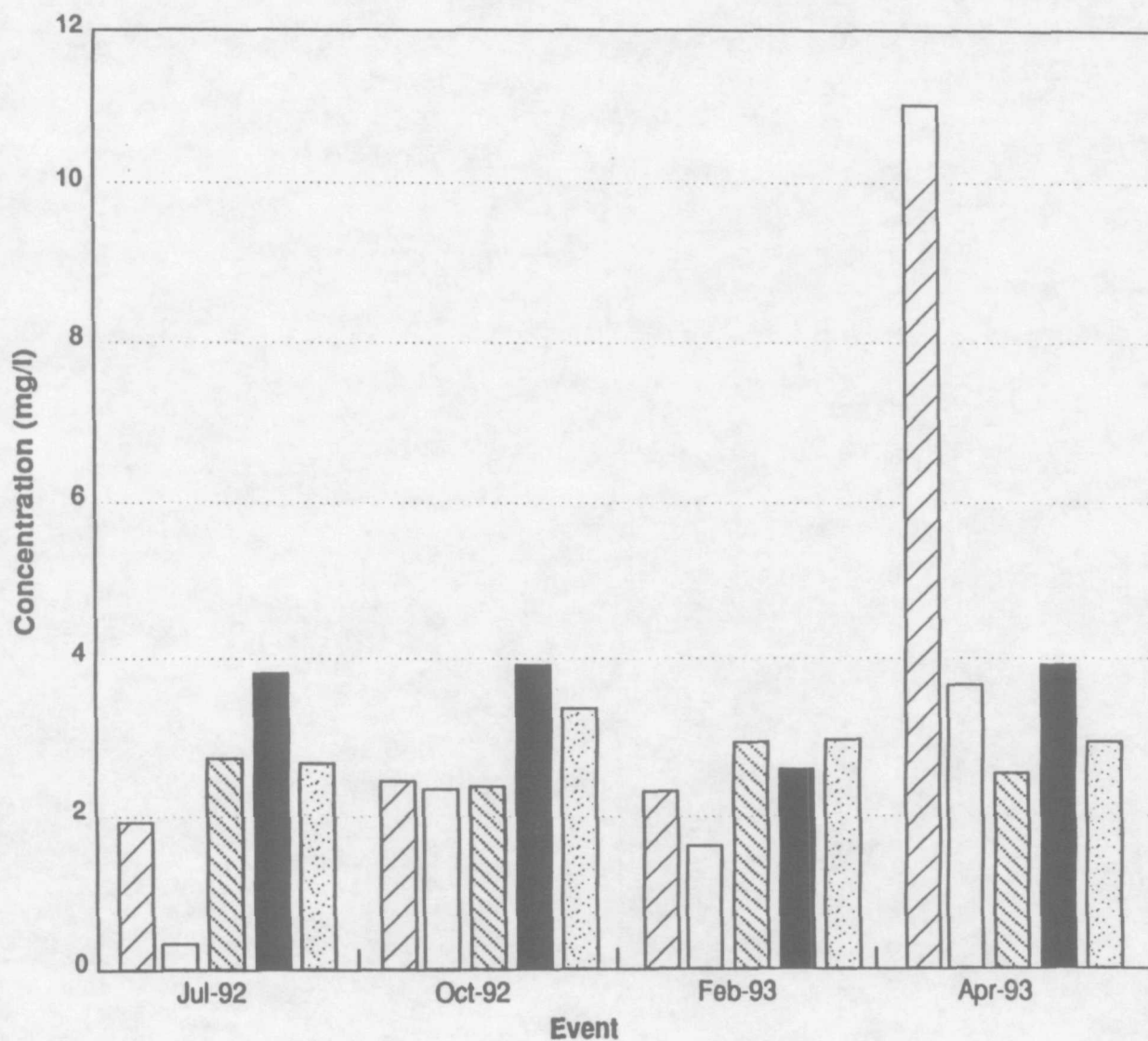
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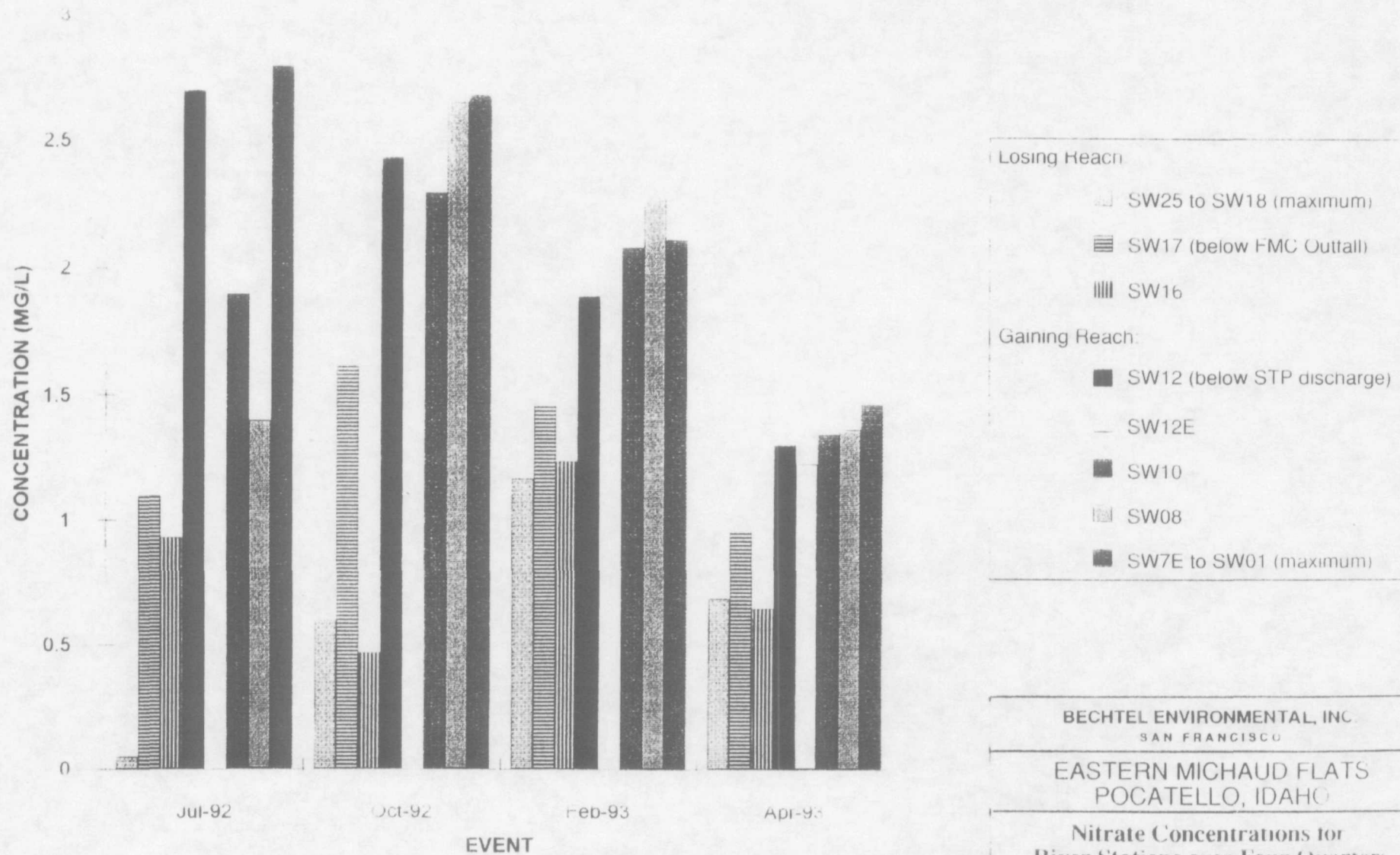
FIGURE 4.5-9

REV.

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BECHTEL ENVIRONMENTAL, INC. SAN FRANCISCO		
EASTERN MICHAUD FLATS POCATELLO, IDAHO		
Nitrate Concentrations for Springs over Four Quarters		
	JOB No.	DRAWING NO.
	21372	FIGURE 4.5-10
		REV.



BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Nitrate Concentrations for
River Stations over Four Quarters



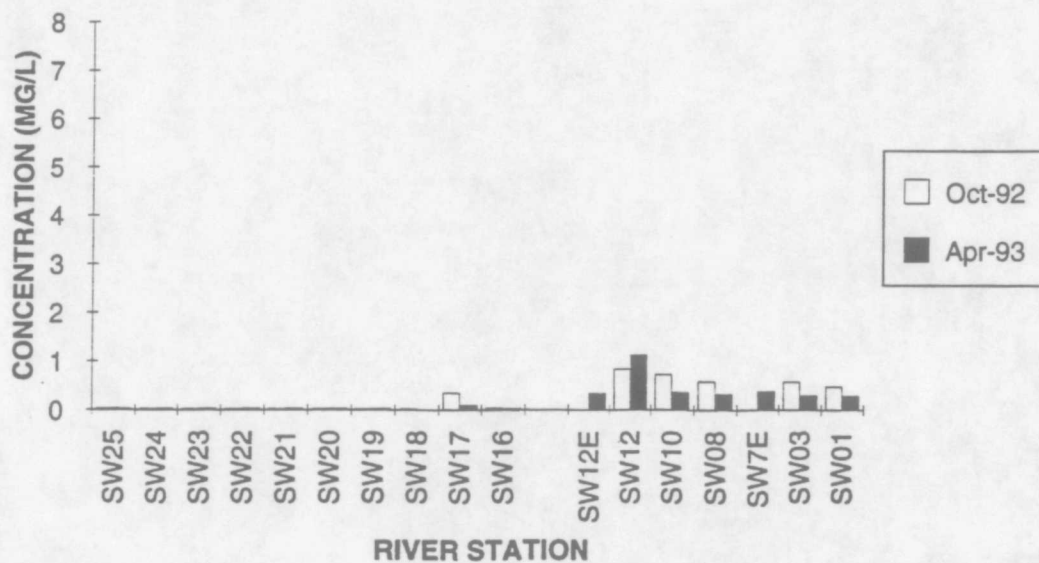
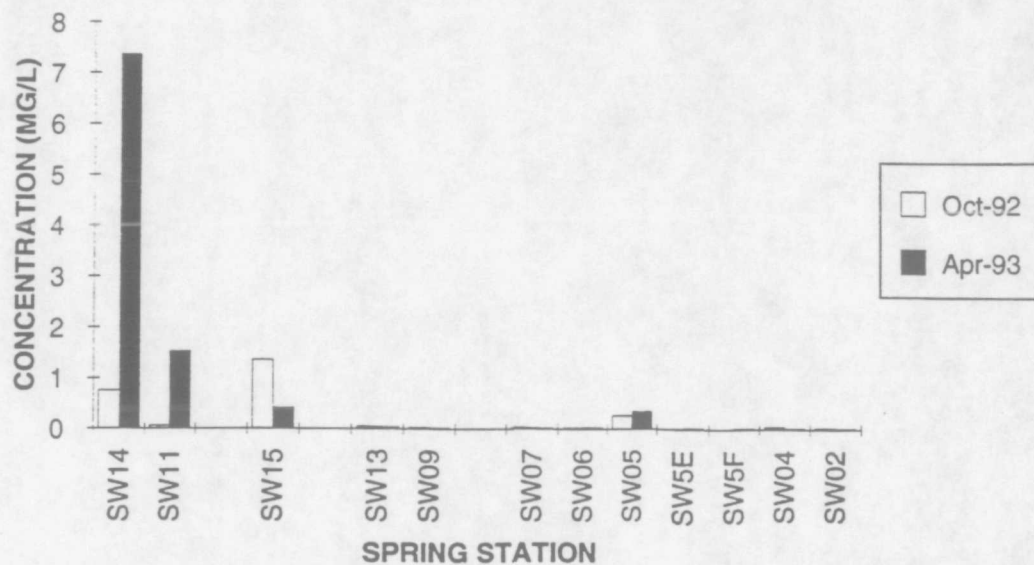
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FIGURE 4.5-11

REV.



BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Orthophosphate in
Springs and River



JOB No.

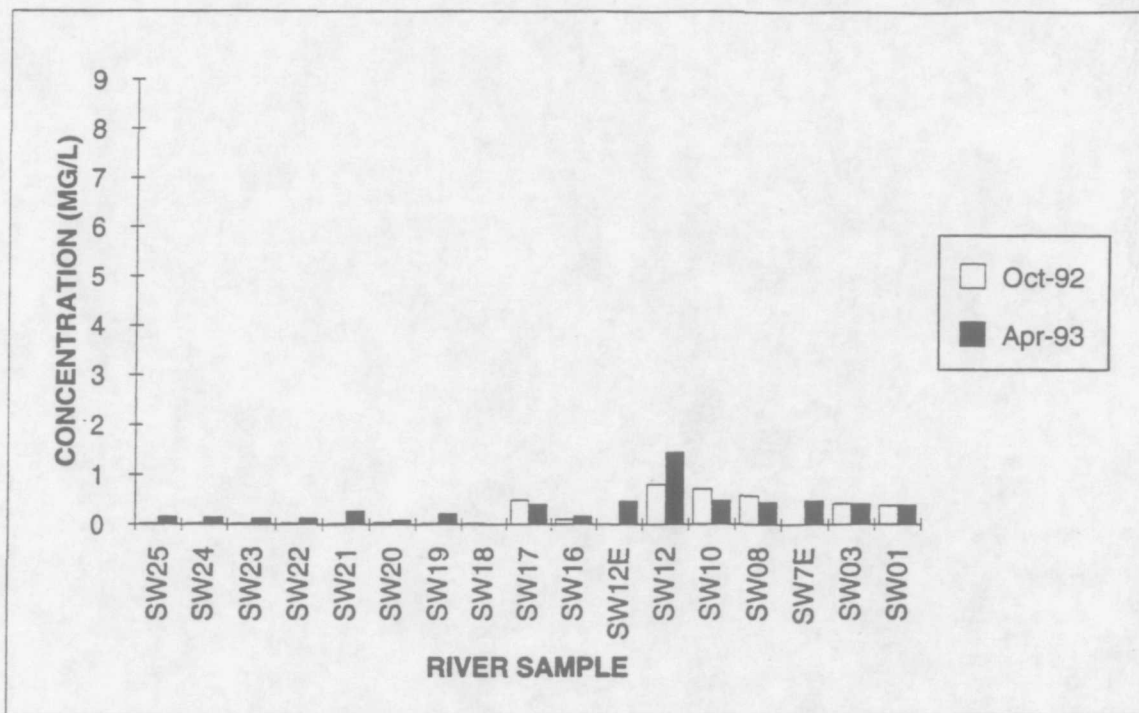
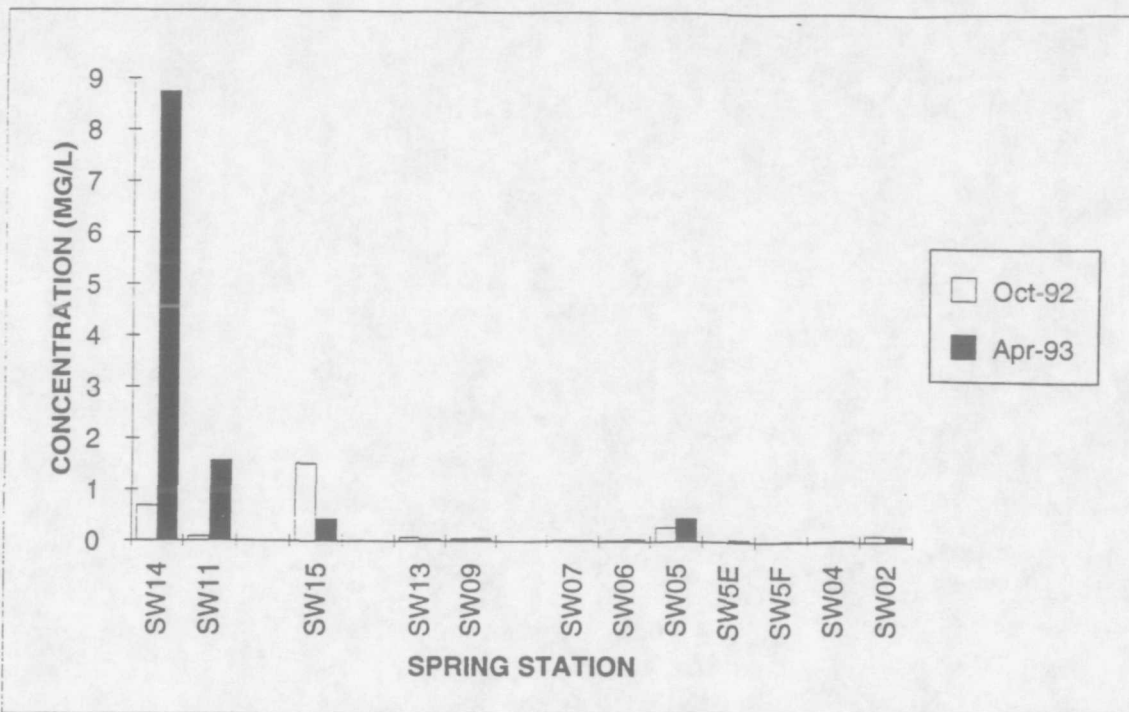
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FIGURE 4.5-12

REV.

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BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Total Phosphorous in
Springs and River



JOB No.

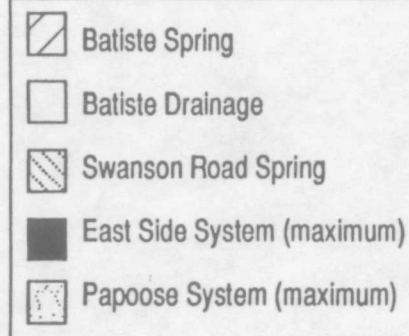
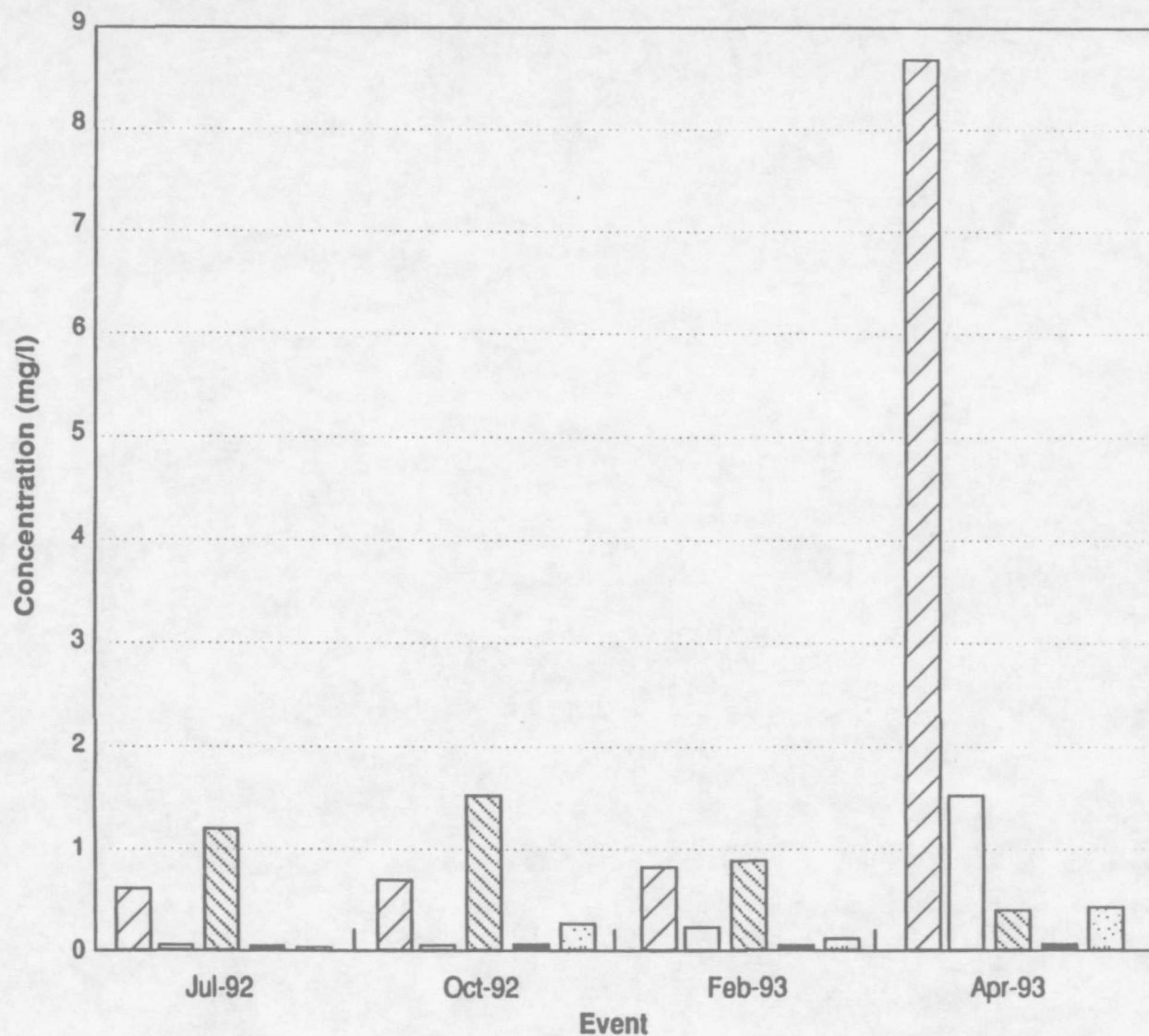
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REV.

21372

FIGURE 4.5 -13

0



BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Total Phosphorus Concentrations for
Springs over Four Quarters



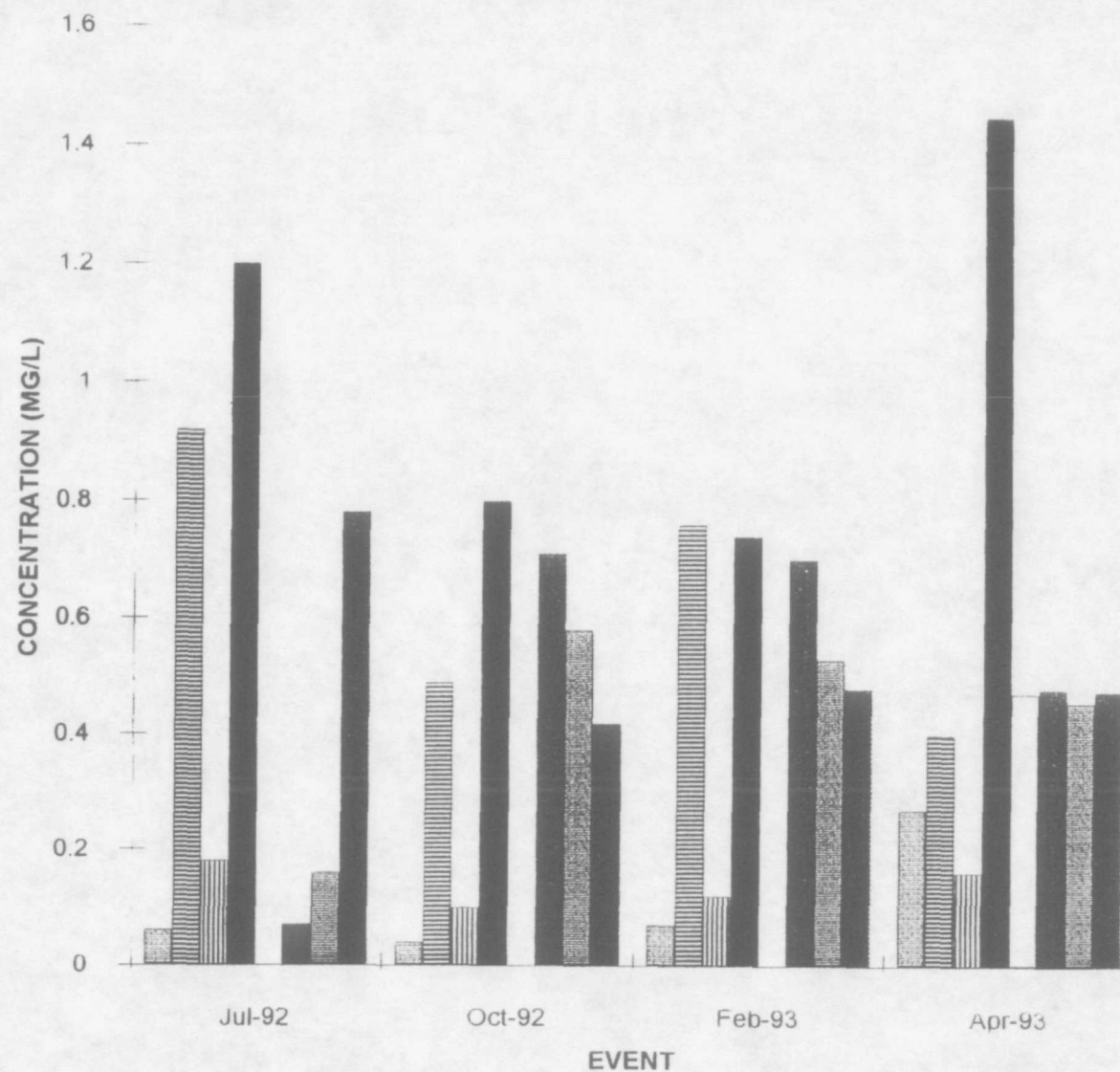
JOB No.

21372

DRAWING NO.

FIGURE 4.5-14

REV.



Losing Reach:

- SW25 to SW18 (maximum)
- SW17 (below FMC Outfall)
- SW16

Gaining Reach:

- SW12 (below STP discharge)
- SW12E
- SW10
- SW08
- SW7E to SW01 (maximum)

BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

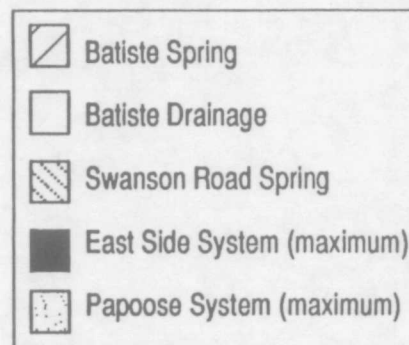
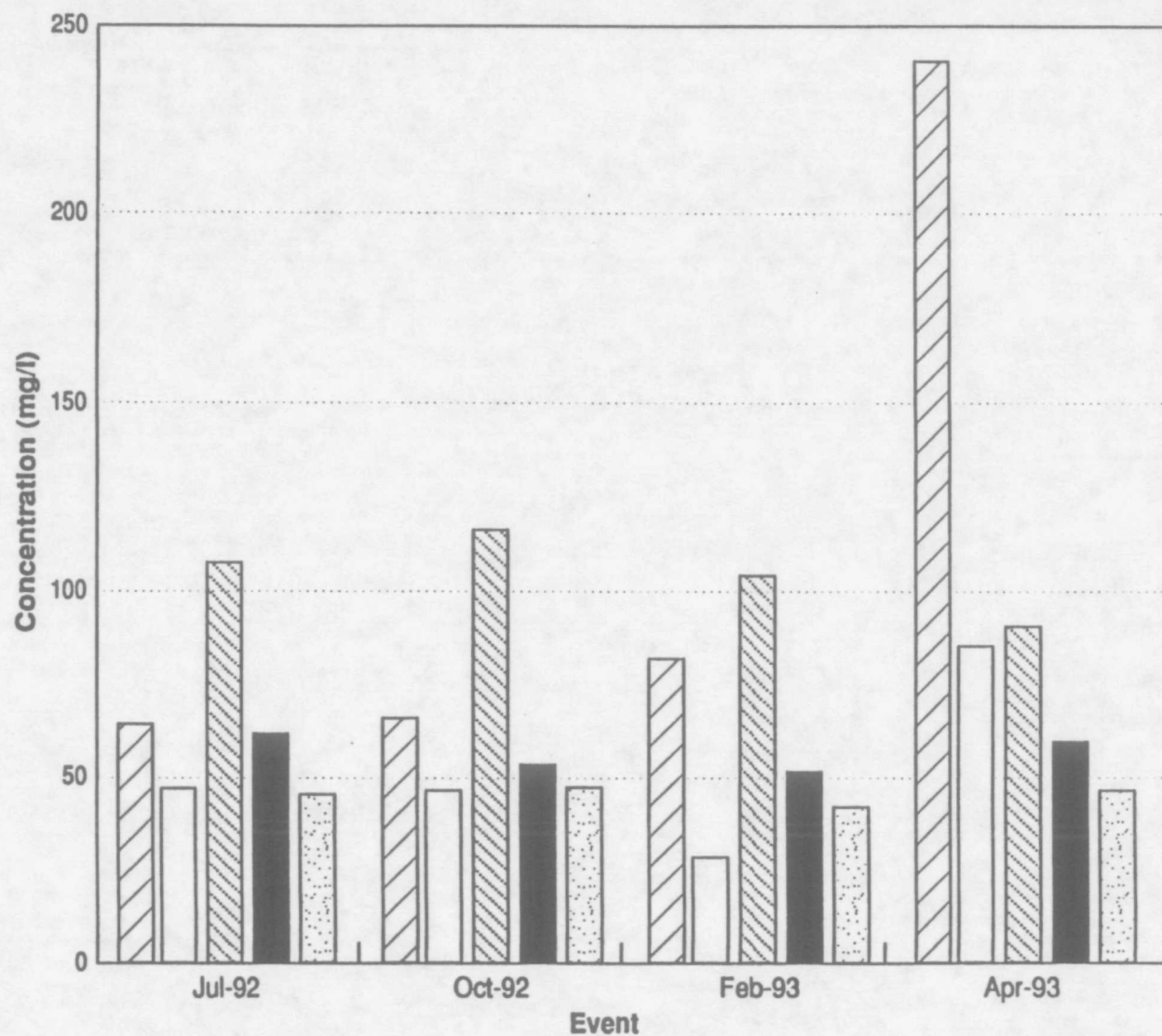
Total Phosphorus Concentrations for
River Stations over Four Quarters



JOB No
21372

DRAWING NO
FIGURE 4-5-15

REV



BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Sulfate Concentrations for
Springs over Four Quarters



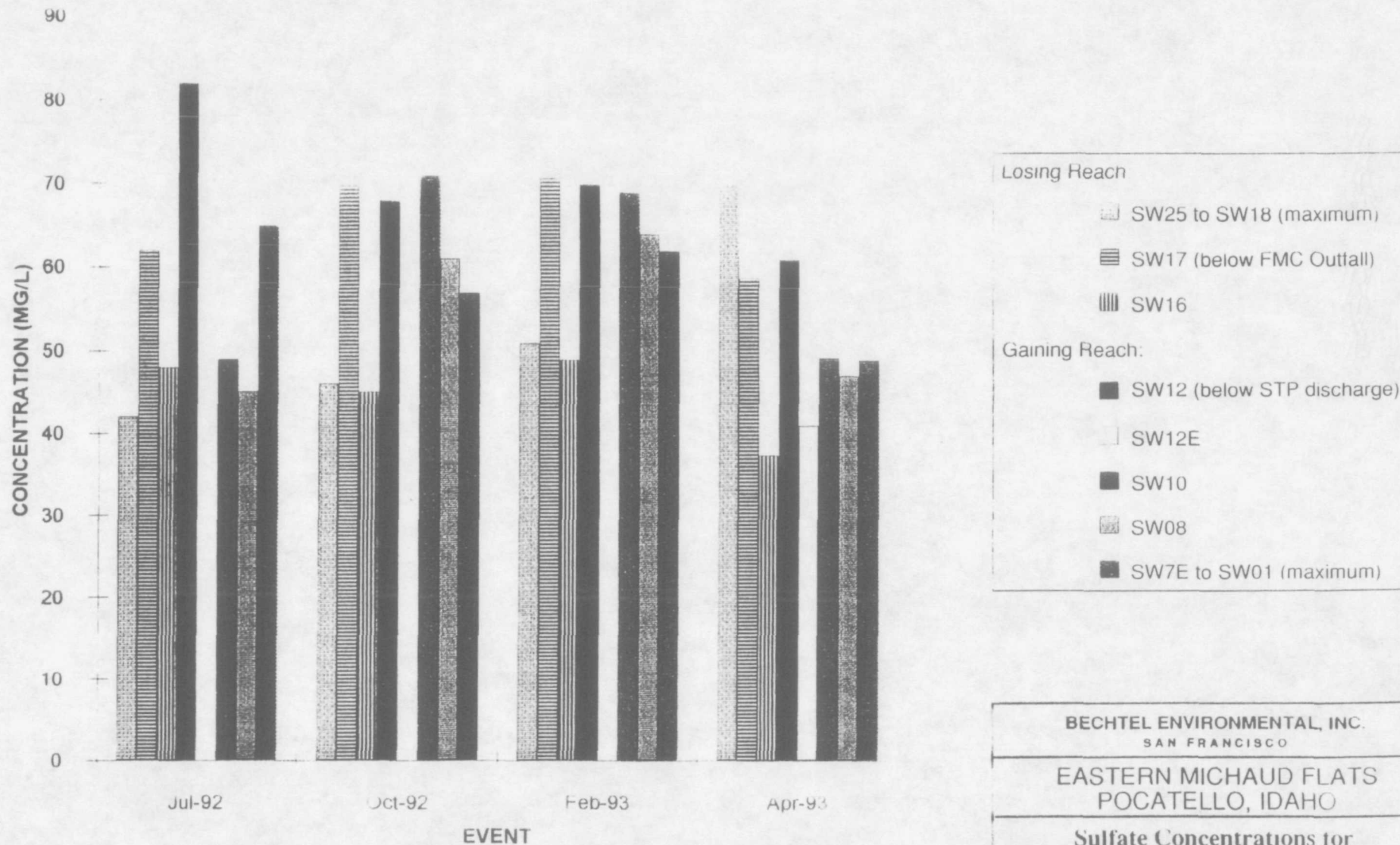
JOB No.

21372

DRAWING NO.

FIGURE 4.5-16

REV.



BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Sulfate Concentrations for
River Stations over Four Quarters



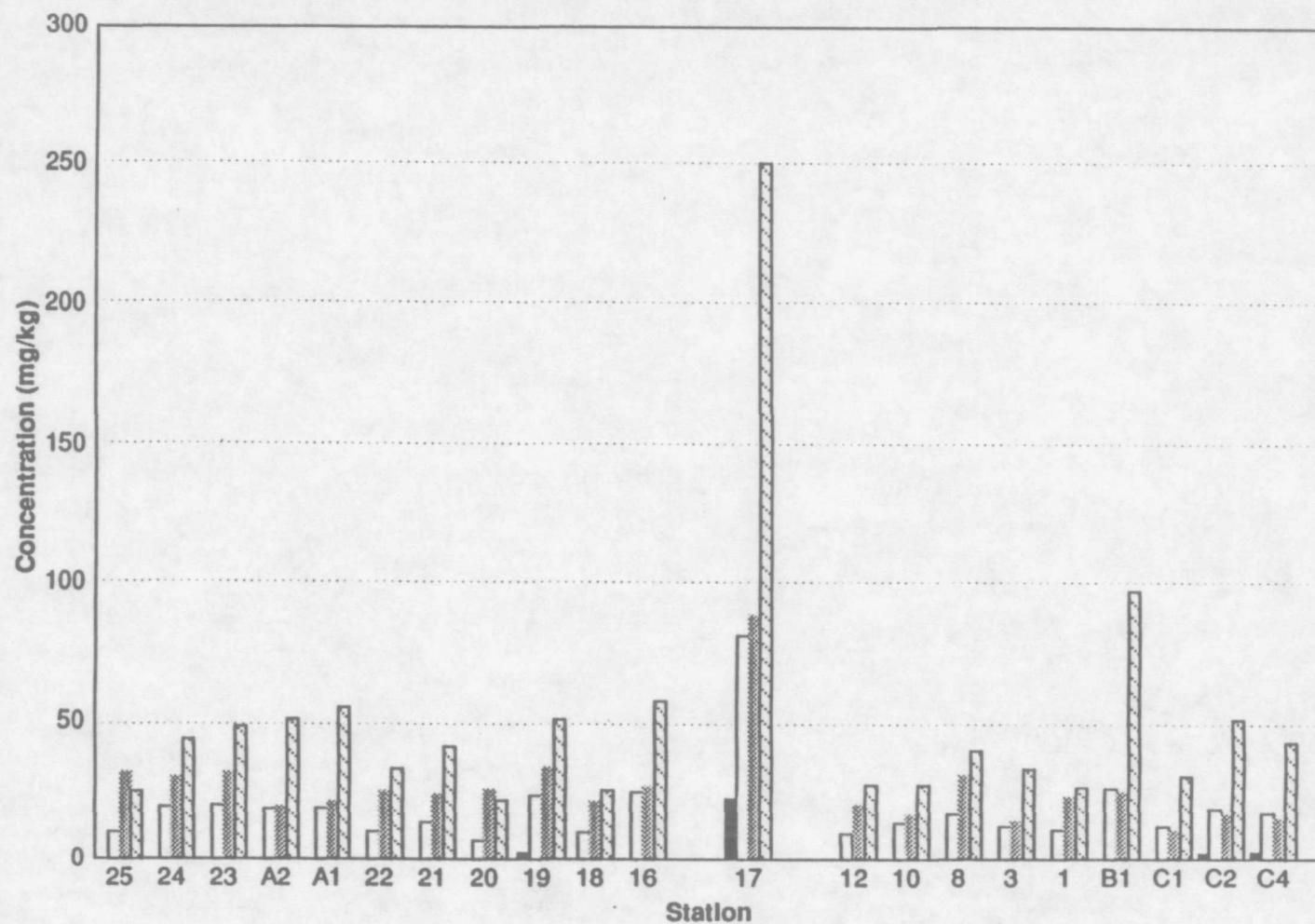
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21372

DRAWING NO

FIGURE 4.5-17

REV



Losing

Gaining

BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Concentrations of Indicator Metals in
Sediments for River Sampling Stations



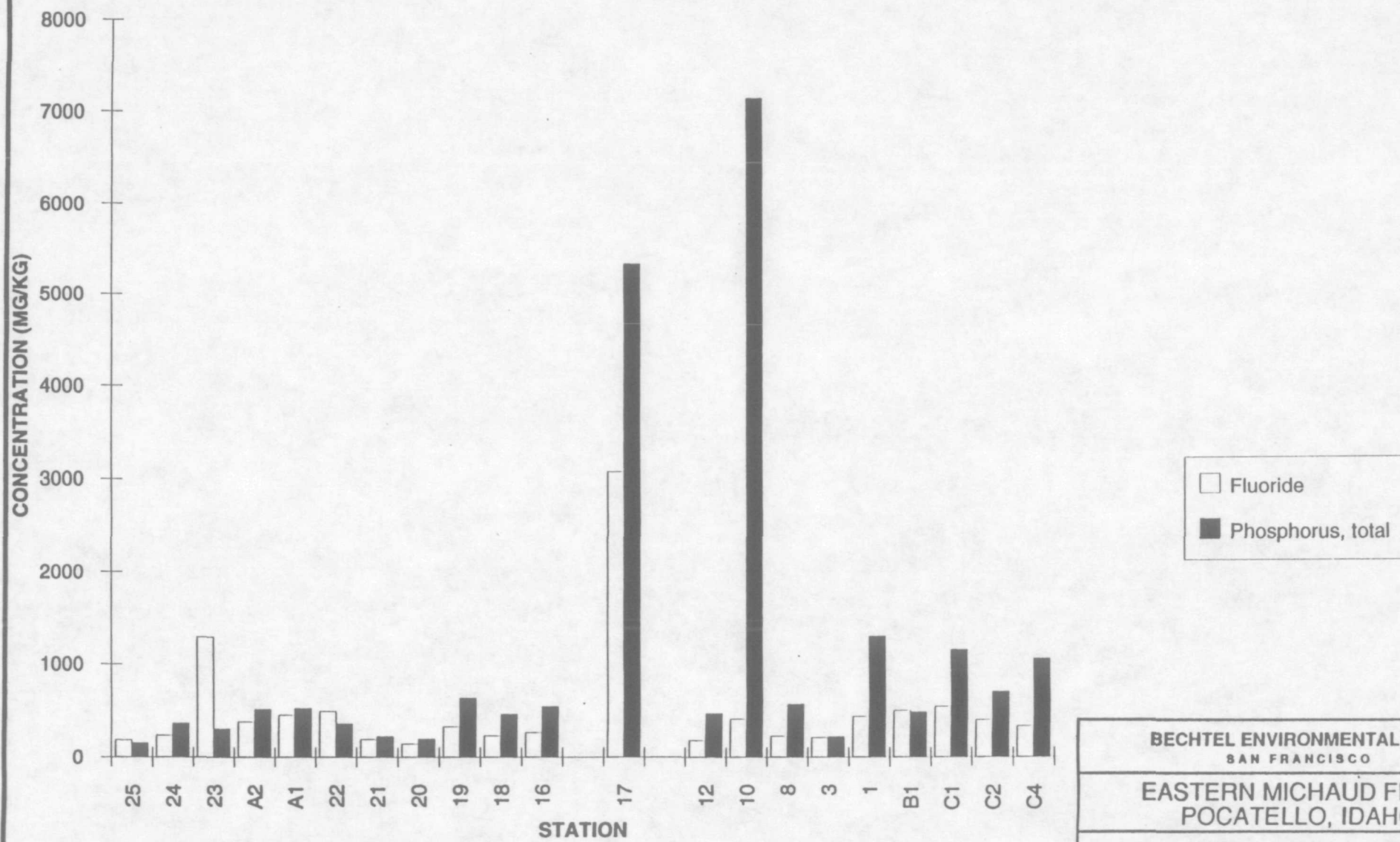
JOB No.


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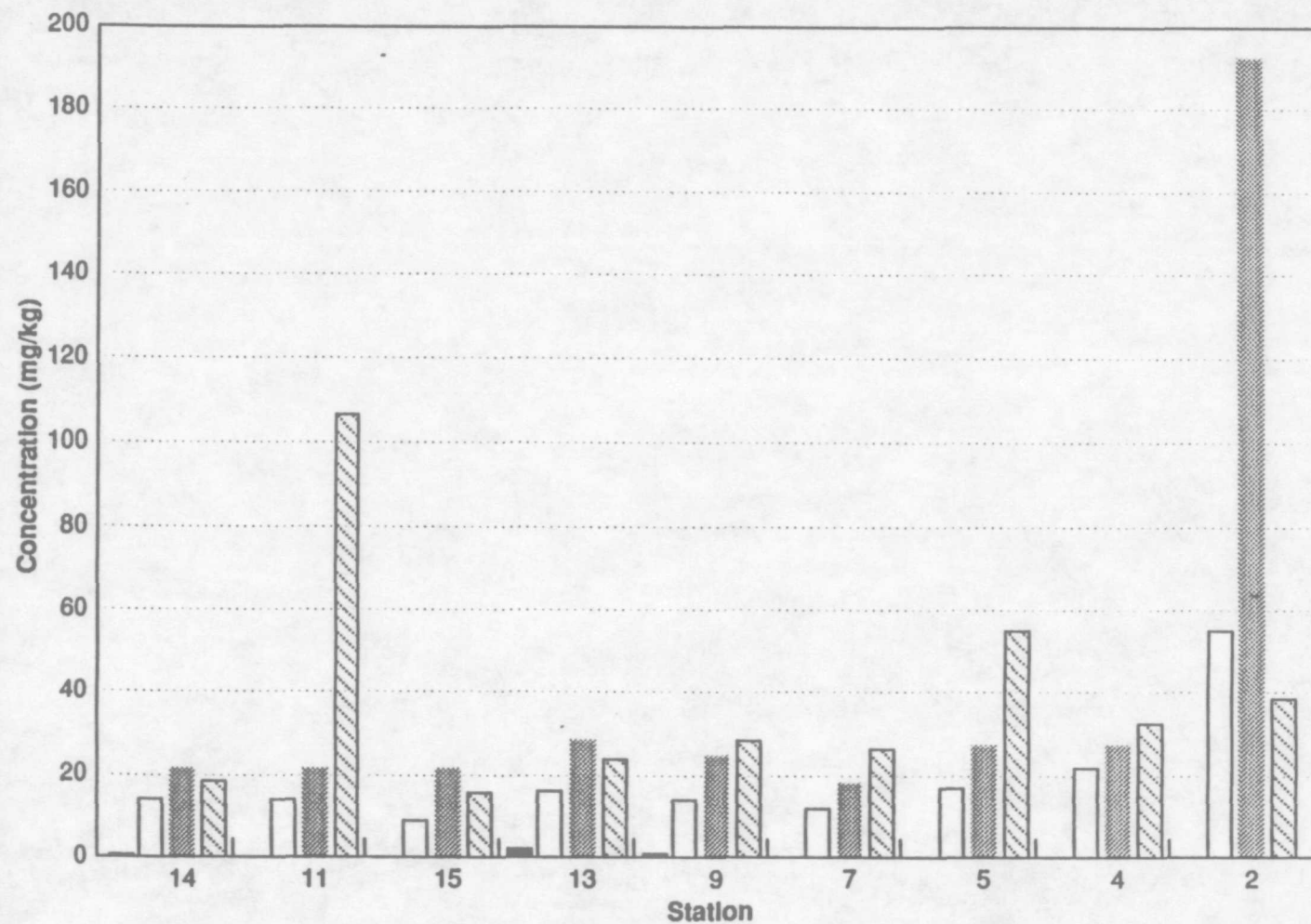
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FIGURE 4.5-18

REV.



BECHTEL ENVIRONMENTAL, INC. SAN FRANCISCO			
EASTERN MICHAUD FLATS POCATELLO, IDAHO			
Concentrations of Total Phosphorus and Fluoride in River Sediments			
	JOB No.	DRAWING NO.	REV.
	21372	FIGURE 4.5.-19	



BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Concentrations of Indicator Metals
in Spring Sediments



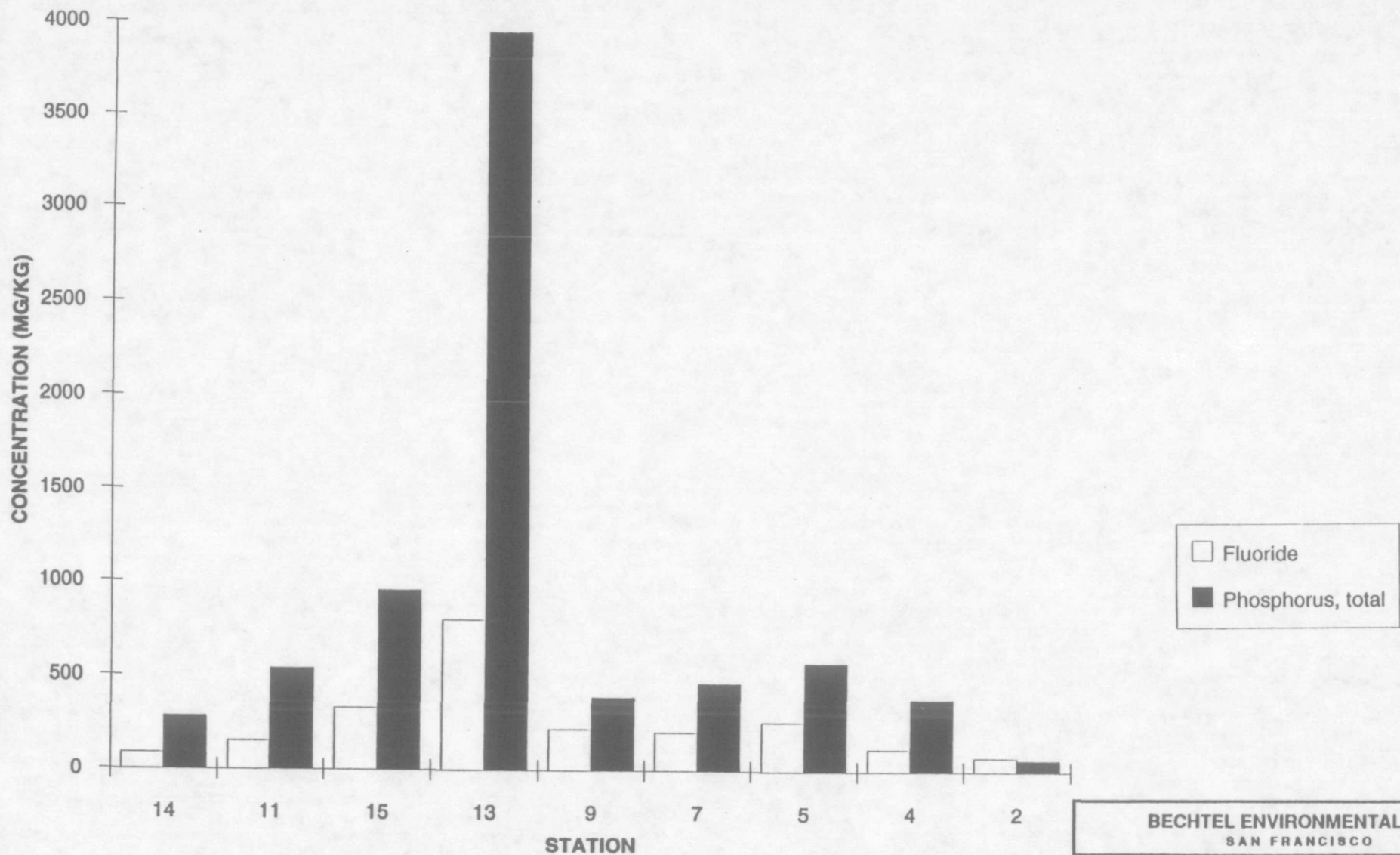
JOB No.

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DRAWING NO.

FIGURE 4.5-20

REV



BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Concentrations of Total Phosphorus
and Fluoride in Spring Sediments



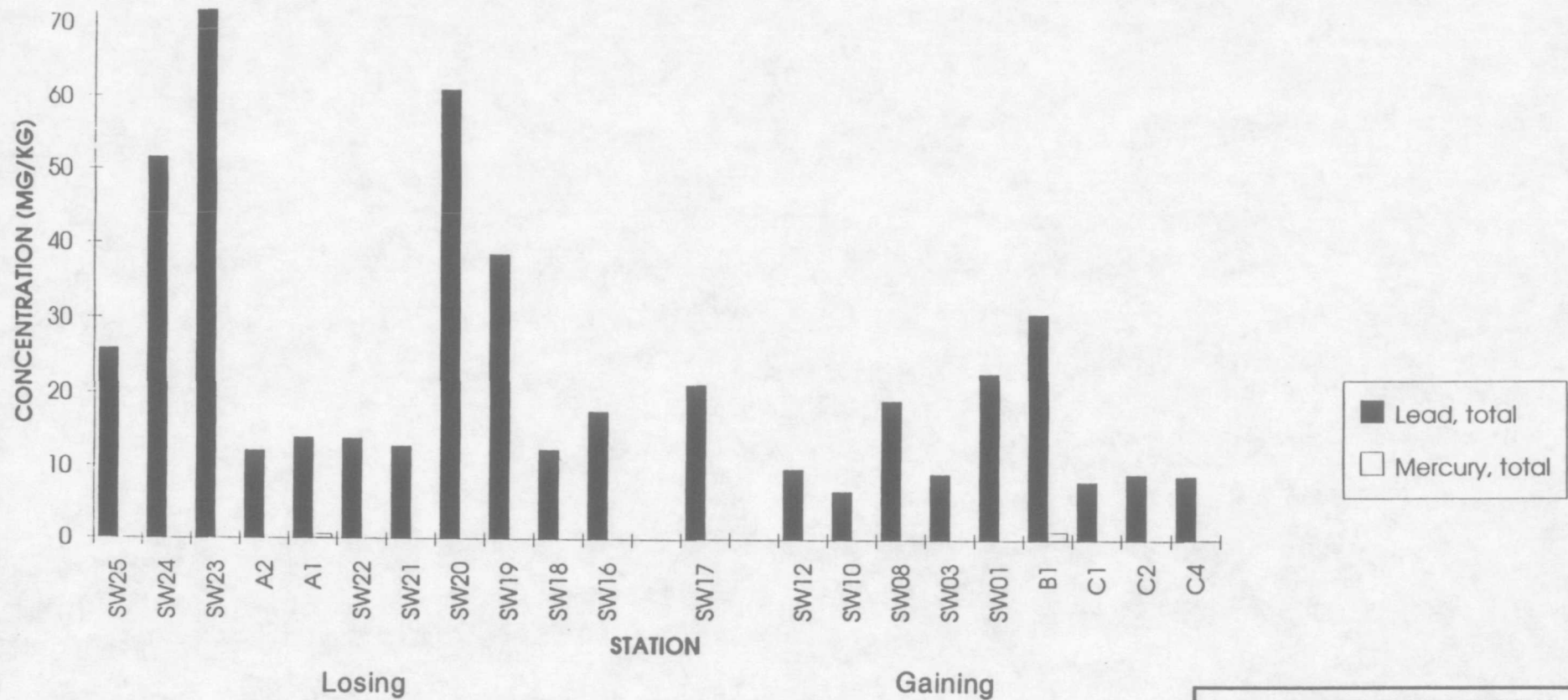
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FIGURE 4.5-21

REV



BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Concentrations of Lead and Mercury in
Sediments for River Sampling Stations



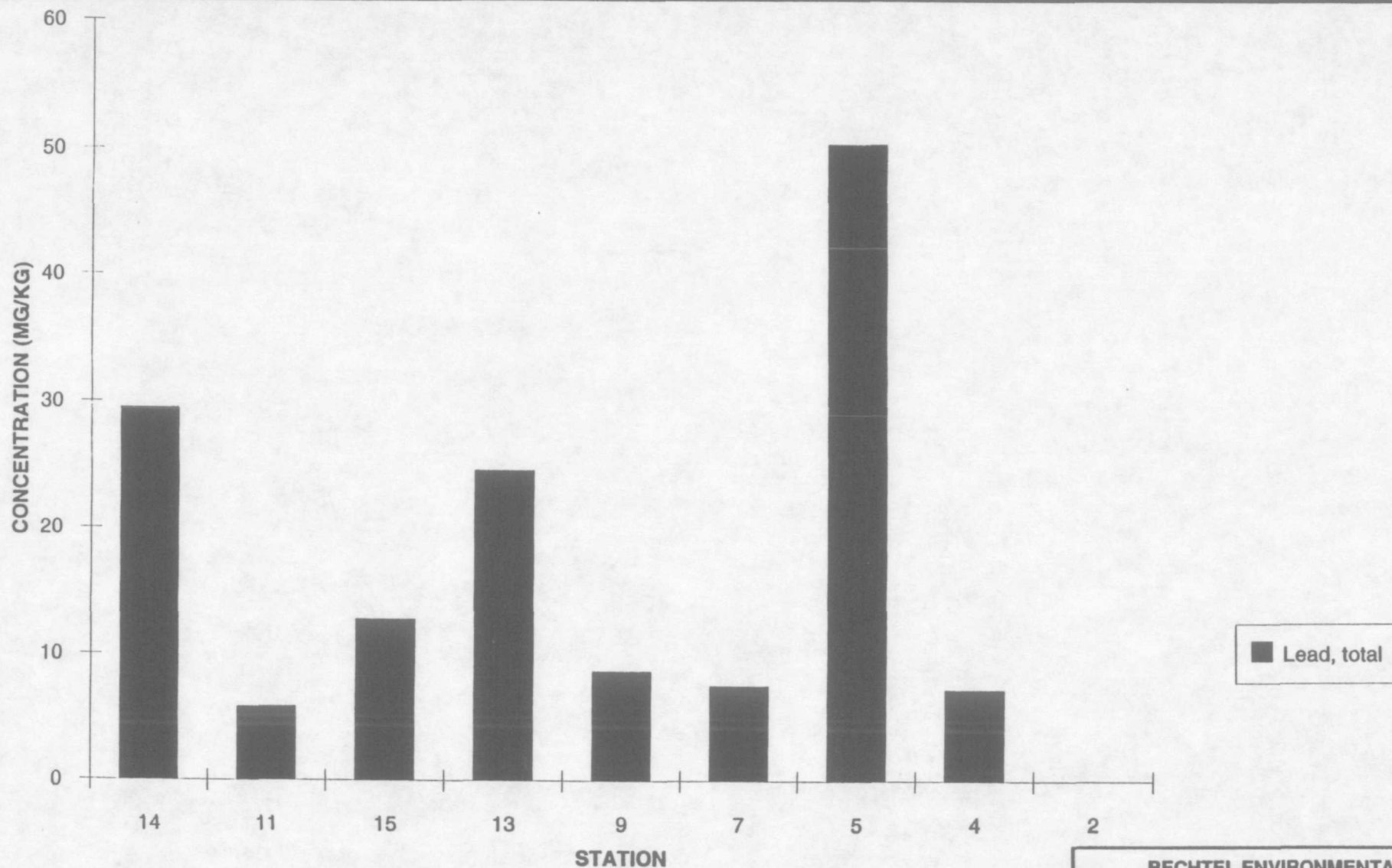
JOB No.


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DRAWING NO.

FIGURE 4.5-22

REV.



BECHTEL ENVIRONMENTAL, INC. SAN FRANCISCO			
EASTERN MICHAUD FLATS POCATELLO, IDAHO			
Concentrations of Lead in Spring Sediments			
	JOB No.	DRAWING NO.	REV.
	21372	FIGURE 4.5-23	

4.6 ECOLOGY

The previous sections discussed the results of the RI/FS SAP (Bechtel, 1992a). The RI/FS SAP was designed to investigate the nature and extent of chemical constituents in air, soils, surface water, and groundwater in the vicinity of the EMF facilities. Despite the findings of these initial studies which indicated that the extent of EMF-related contamination is restricted to the close proximity of the facilities, EPA remained concerned about potential impacts to local ecosystems. Thus, an ecological investigation was undertaken at EPA's request to provide additional site-specific data for assessing potential ecosystem impacts associated with EMF activities.

For the purpose of the ecological investigation, only inorganic chemicals were selected as COPCs. Cadmium, fluoride, and zinc were selected as COPCs for both the aquatic and terrestrial investigations by EPA. In the aquatic environment, arsenic, mercury, and selenium, in addition to cadmium, fluoride, and zinc, were also identified as COPCs. Arsenic was included because of concern that the EMF facilities contribute this constituent to groundwater discharging to the Portneuf River in the vicinity of Batiste Spring and Swanson Road Spring. Selenium and mercury were included because of their potential to bioaccumulate in the aquatic food chain.

The aquatic and terrestrial ecosystem investigations were performed separately. The aquatic investigation was performed in October, 1994; whereas the terrestrial field sampling effort was conducted in September, 1994. The Ecological Field Sampling Plan (EFSP) Appendix R and Ecological Quality Assurance Project Plan (EQAPjP) (Bechtel, 1994l), which together comprise the Ecological Sampling and Analysis Plan (ESAP), contain a detailed description of the sampling and analytical methodology. An overview of the sampling protocol and analyses, as well as modifications to the EFSP and EQAPjP, were provided in Section 2.7.

ECOLOGICAL INVESTIGATION OBJECTIVES

The objectives of the ecological investigations were:

Aquatic

- To determine the potential impact of EMF-related discharges to the Portneuf River benthic community in the immediate vicinity of FMC's IWW ditch outfall.
- To determine differences between Portneuf River delta sediment COPC concentrations and sediments in both a reference area (Snake River delta) and upstream of the EMF facilities.
- To determine the potential biological impact of sediment COPC concentrations in the Portneuf River delta.

Terrestrial

- To determine differences in the COPC concentrations of soils, vegetation, and small mammals between a reference, non-impacted sagebrush habitat location and two potentially impacted sagebrush habitat locations.
- To determine differences in the COPC concentrations of soils and vegetation between a reference, non-impacted riparian habitat and a potentially impacted riparian habitat location.
- To evaluate the transfer of COPCs into the food chain to assess potential terrestrial ecosystem impacts.

OVERVIEW OF FINDINGS

Aquatic Investigation

- Cadmium was the only COPC detected at an elevated concentration in a sediment sample collected in the immediate vicinity of the IWW ditch outfall (SD17) during this investigation. The sampling location immediately downstream of SD17 (SD16) did not contain elevated cadmium concentrations. In addition, bioassays conducted on sediment

collected from near the outfall at SD17 revealed that cadmium was not toxic to two species of benthic organisms.

- Cadmium was the only COPC elevated in Portneuf River delta sediments, compared to both Snake River delta and upstream Portneuf River sediment samples. However, the Portneuf River delta cadmium concentrations were below levels of ecological concern (LECs) established by sediment bioassays, below representative soil levels, and below the concentration detected at SD17 (adjacent to the IWW ditch outfall) for which no toxicity was found.

Terrestrial Investigation

- Most cadmium, fluoride, and zinc concentrations were increased at potentially impacted soil sampling locations, relative to reference soils.
- Vegetation cadmium and fluoride concentrations from potentially impacted areas were elevated compared to reference location areas; zinc concentrations were not elevated.
- Washed and unwashed vegetation uptake parameters for cadmium and zinc were an order of magnitude lower than default uptake values provided by Baes et al. (1984). This finding may be attributable to the high pH of soils in the EMF study area and the lower bioavailability of EMF-related constituents. Although washed vegetation fluoride concentrations were not available, a comparison of unwashed vegetation levels with soil concentrations indicated less plant uptake than would be predicted using the Baes model for sagebrush and wheatgrass.
- Tissue concentrations of cadmium and fluoride were elevated in mice captured from potentially impacted locations, relative to reference areas. These concentrations were below levels of ecological concern with respect to any potential impact to mice or other small mammals.
- The mouse cadmium and zinc tissue concentrations were well below levels associated with toxicity in top predators who prey on mice and other small mammals. Fluoride intoxication in top predators is unlikely given data on mouse tissue levels.
- Cadmium, fluoride and zinc vegetation concentrations are not high enough to result in adverse effects in a representative ruminant, the mule deer, due to the limited exposure associated with the daily and seasonal movement activities of these ruminants.

4.6.1 AQUATIC INVESTIGATION

Two separate aquatic investigations were performed as described in the ESAP (summarized in Section 2.7.1). The aquatic investigations consisted of: 1) an assessment of the aquatic environment near the FMC's IWW ditch outfall; and 2) a Portneuf River Delta Study undertaken at the confluence of the Portneuf River and the American Falls Reservoir. The results of these analyses provide an assessment of the potential nature and extent of contamination with respect to ecological receptors. Although this section does not specifically address the potential ecological impacts, available information on levels of ecological concern are referenced where appropriate.

4.6.1.1 IWW Ditch Outfall Investigation

As described in Section 2.7.1.1, three composite sediment samples were collected in the vicinity of the IWW ditch outfall in September 1994 as part of the ecological investigation. The sampling of sediment in the vicinity of the IWW ditch outfall for the ecological investigation was prompted by concern over the impact of site-related discharges to the benthic community in the immediate vicinity of the outfall. As detailed in Section 4.5, the sediment sampling location located immediately downstream of the IWW ditch outfall (SD17) is the only location at which EMF-related influences were measurable during previous investigations. Consequently, 10-day toxicity tests, using the organisms *Hyallela* and *Chironomous*, were conducted for sediments collected near sampling location SD17 (Figure 4.5-1f). In addition, sediments were collected from two other nearby sampling locations for chemical and toxicity analyses. The other sampling locations were: 1) upstream of the IWW ditch outfall (SD21), and; 2) downstream of the IWW ditch outfall sampling location SD17 (SD16).

Each collected sediment sample was analyzed for specific physical and chemical parameters including simultaneously extractable metals (SEM)/acid volatile sulfide (AVS), nutrients, total organic carbon (TOC), TAL metals and fluoride. Five of the TAL metals (arsenic, cadmium, mercury, selenium, and zinc) and fluoride were considered as COPCs by EPA. Cadmium, zinc

and fluoride were selected as COPCs as these constituents have been detected at elevated levels adjacent to the EMF facilities (Section 4.3). Arsenic was included due to the agency's concern that EMF-related activities have potentially resulted in contributions of this constituent to the river. Selenium and mercury were included as COPCs because of their potential to bioaccumulate.

Physicochemical Parameters of Sediment and Surface Water Recorded at Sampling Stations SD16, SD17, and SD21.

The IWW ditch outfall investigation sampling locations corresponded with locations sampled during the previous RI sediment investigations; the chemical parameters recorded at each sampling station during these previous investigations are fully described in Section 4.5.2.1. Only the data obtained as a part of the ecological studies are described in this section.

Stations SD16 and SD17 were characterized by a rocky bottom, making the collection of fine sediments difficult. While the river bed at station SD21 was also composed of gravel and small cobbles, these were covered with a layer of silt. At all stations at least three samples, representing transects across the river, were taken and composited.

The levels of most surface water physicochemical parameters were similar at all three sampling stations (Table 4.6-1). However, dissolved oxygen and turbidity were higher at station SD21, upstream of the IWW ditch outfall. In addition, water flow was significantly faster at stations SD16 and SD17 than at station SD21.

Chemical Analysis and Bioassays of Sediment COPC Concentrations During the IWW Ditch Outfall Investigation

For reasons detailed above, the COPCs analyzed in sediments during the IWW ditch outfall investigation consisted of fluoride and the TAL metals arsenic, cadmium, mercury, selenium and zinc. As called for in the ESAP, the full list of TAL metals were analyzed for but, ultimately, constituents other than arsenic, cadmium, mercury, selenium and zinc were not considered as

COPCs. The results of the sediment analyses at sampling stations SD16 (downstream of the IWW ditch outfall), SD17 (at the outfall), and SD21 (upstream of the outfall) are summarized in Table 4.6-1.

For discussion purposes, representative sediment levels upstream of the IWW ditch outfall have been developed based on upstream sediment data collected from sample locations SD21, SD22, SD23, SD24, SD25, SDA1 and SDA2 during previous phases of the RI (discussed in Section 4.5). Provided a COPC was detected in over 50 percent of the upstream sediment samples, the mean COPC concentration was calculated for comparison to the concentrations detected in the IWW ditch outfall samples. For non-detect samples, one half of the sample quantitation limit was used in the calculation of mean upstream COPC concentrations. For COPCs that were detected in less than 50 percent of the upstream samples, a mean upstream concentration was not deemed suitable for use in the analysis and was, therefore, not calculated. A comparison of upstream data to sediment concentrations detected during the IWW ditch outfall investigation is presented in Table 4.6-2.

In addition to comparison of upstream values, concentrations detected during the IWW ditch outfall investigation were compared to levels of ecological concern (LECs) (Table 4.6-3). EPA has suggested that the lowest effects levels (LELs) established in the Province of Ontario's Ministry of Environment and Energy (OMEE) sediment guidelines (Persaud et al., 1993) be used as LECs. The OMEE LELs are intended to represent an effects-based approach using the presence (or absence) of aquatic species as endpoints.

In essence, the approach used to establish the OMEE LELs assumes that the absence of one (or more) species or genus from a pre-defined set of 100 species or genera is directly attributable to the presence of an elevated single contaminant concentration in the site sediment. By making this assumption, the authors ignore the impact attributable to co-contaminants or, perhaps more importantly, habitat quality as related to distinct surface water and sediment characteristics. As a result, the LELs established by Persaud et al. (1993) are not based on a causative relationship

between contaminant concentration and biological effect. In fact, some of the LELs are actually above pre-industrial background concentrations, established from core sediment dating techniques. For example, the pre-industrial Great Lakes sediment cadmium concentration (1.1 mg/kg) is nearly twice as high as the OMEE LEL (0.6 mg/kg). Based on these findings, it was determined that the OMEE sediment LELs are not suitable for use as LECs in the analysis of the results of the IWW ditch outfall investigation.

At the current time, EPA's Region 10 is considering developing effects-based sediment guidance for certain chemical contaminants (Cubbage and Breidenbach, 1994) which would be more suitable as LECs. This proposed guidance is based on the determination of apparent effect toxicity thresholds (AETs) in sediment bioassays of individual contaminants. AETs have been calculated for both Microtox (an aquatic bacterial assay of general toxicity) and in the aquatic organism *Hyallela*. Because they are primarily based on a measurement of toxicological effects in aquatic organisms, the Region 10 AETs provide a better basis for sediment guidelines than the arbitrarily determined OMEE LELs. Consequently, these values were selected for use as LECs in the IWW ditch outfall investigation. However, the LELs derived by the OMEE (Persaud et al., 1993) are included in Table 4.6-3 and the following discussion for completeness. In addition, representative soil levels (Section 4.2) are also presented for comparison. The LECs, as well as the limitations of the OMEE LELs, are discussed in greater detail within the report "The Determination of the Need to Proceed with Phase II of the River Delta Study" (SII, 1994). In presenting the results of the investigation, each COPC is discussed separately.

Arsenic. The average arsenic concentration in the two downstream IWW ditch outfall investigation samples was 2.45 mg/kg, with a maximum of 2.5 mg/kg immediately downstream of the IWW ditch outfall (station SD17); the upstream value (station SD21) was 2.4 mg/kg. These concentrations are below the LEC of 40 mg/kg, which is based on the lowest AET observed during freshwater bioassays conducted by EPA Region 10 (Cubbage and Breidenbach, 1994). The concentrations are also below the OMEE LEL of 6 mg/kg (Persaud et al., 1993), and

upstream concentrations (associated with previous RI studies) which average 4.3 mg/kg. Consequently, the IWW ditch outfall does not appear to contribute significant concentrations of arsenic to sediments in the Portneuf River.

Cadmium. The highest concentration of cadmium detected in sediments during the IWW ditch outfall investigation (3.2 mg/kg) occurred in the immediate vicinity of the outfall at Station SD17. This concentration is below the LEC of 7 mg/kg, developed based on the lowest AET observed during Region 10 freshwater bioassay research (Cubbage and Breidenbach, 1994). However, it exceeds the OMEE LEL (Persaud et al., 1993) guidelines (0.6 mg/kg), the representative soil level (1.9 mg/kg) and the maximum cadmium concentration detected in upstream sediments during previous RI studies (0.82 mg/kg).

Sediments collected immediately downstream (SD16) and upstream (SD21) were found to contain 0.7 mg/kg and 0.44 mg/kg of cadmium, respectively. Neither value exceeds the LEC of 7 mg/kg. While the sample collected at SD16 does exceed the OMEE LEL, it is below the maximum upstream concentration detected during previous RI sediment studies (0.82 mg/kg at SD21).

In summary, while effluent from the IWW ditch outfall has resulted in elevated cadmium concentrations in sediment located in the immediate vicinity of the outfall, these concentrations do not exceed a level of ecological concern.

Fluoride. A fluoride LEC could not be developed as Region 10 has not derived an AET for this chemical. In addition, OMEE has not derived an LEL for fluoride. However, fluoride concentrations in each of the IWW ditch outfall sediment samples (183-312 mg/kg) were below the mean upstream concentration detected during previous RI studies (330 mg/kg), and below the representative soil level (600 mg/kg). Consequently, any discharge of fluoride from the IWW ditch outfall into the Portneuf River has not had a significant effect on sediment quality in the immediate vicinity of the outfall.

Mercury. Mercury was not detected at any of the three IWW ditch outfall sediment sampling stations. Quantitation limits ranged from 0.025 to 0.03 mg/kg. These quantitation limits are well below the Region 10 LEC (0.56 mg/kg), the OMEE LEL (0.2 mg/kg). In addition, the quantitation limits are well below the maximum upstream concentration of mercury detected during previous RI studies (0.55 mg/kg at SDA1). In summary, the IWW ditch outfall does not appear to be contributing significant amounts of mercury to sediments within the Portneuf River.

Selenium. An LEC for selenium could not be developed as Region 10 has not derived an AET for this chemical. In addition, OMEE has not derived an LEL for selenium. However, selenium was not detected at any of the three IWW ditch outfall sampling stations. Quantitation limits ranged from 0.14 to 0.18 mg/kg. These values are well below previously detected selenium concentrations in upstream samples (0.54 mg/kg and 0.72 mg/kg at sample locations SD24 and SD25, respectively). Therefore, the IWW ditch outfall does not appear to be contributing significant amounts of selenium to Portneuf River sediments.

Zinc. Zinc sediment concentrations ranged from 18.3 to 49 mg/kg. These concentrations are all either below or in the range of detected upstream concentrations (24.3-55.3 mg/kg). In addition, they are all at least an order of magnitude lower than the LEC (490 mg/kg) (Cubbage and Breidenbach, 1994). Finally, these concentrations are all below both the OMEE LEL (120 mg/kg) and the representative soil level (52.8 mg/kg). In summary, the zinc sediment levels are in the range of representative upstream concentrations and below a level of ecological concern.

Results of sediment bioassays

As described in Section 2.7, sediment samples collected at SD16, SD17 and SD21 were analyzed for 10-day survival studies using *H. azteca* and *C. tentans* as test organisms. The results of these studies indicated no adverse effects on either survivability or growth in either species in any of the sediments collected as would be expected based on the LECs (Table 4.6-4).

Summary of the IWW Ditch Outfall Ecological Investigation

A qualitative analysis of sediments collected at stations SD16, SD17, and SD21 did not indicate site-related impacts to aquatic habitats in the Portneuf River in the vicinity of the IWW ditch outfall. The chemical analysis of sediments revealed that arsenic, fluoride and zinc are present in sediments immediately downstream of the IWW ditch outfall at concentrations within the range detected in upstream, non-EMF impacted samples. By contrast, mercury and selenium were not detected in any of the IWW ditch outfall Investigation samples. Detection limits for both of these chemicals were below the maximum concentration detected in non-EMF impacted, upstream samples. Finally, while cadmium was detected at a concentration above all previous upstream concentrations in the immediate vicinity of the IWW ditch outfall (SD17), it did not exceed the LEC.

The sediment analyses indicate that no impact is expected either at or downstream of the IWW ditch outfall. These findings were verified by the results of toxicity tests, conducted with the same three bulk sediment samples, that did not show any adverse impact on macroinvertebrate test organisms during 10-day bioassays.

4.6.1.2 Portneuf River Delta Study

The Portneuf River Delta Study, summarized in Section 2.7 and described in detail in the ESAP (Appendix R), consisted of two phases. Phase II was to proceed only if the Phase I sediment analyses of COPCs indicated that a further investigation was warranted based on a set of triggering criteria associated with COPC delta sediment concentrations. The decision tree used to determine whether a Phase II study should be completed is described in detail in Appendix S. The six COPCs consisted of arsenic, cadmium, fluoride, mercury, selenium, and zinc. These constituents were selected for the same reasons as detailed in Section 4.6.1.1. Based on the results of Phase I, which are presented in this section, Phase II was determined to be unnecessary.

The purpose of the Phase I Portneuf River Delta Study was to determine whether potential EMF related COPCs exist within the river delta sediments at concentrations significantly greater than representative levels and, if so, whether the levels in the Portneuf River delta sediments pose a hazard to aquatic biota. The representative levels were developed based on sediment data collected from both the Snake River delta and the Portneuf River upstream of the EMF facilities. The potential hazard to aquatic biota was assessed using the same LECs adopted for the IWW ditch outfall investigation. A comparison to the LELs derived by the OMEE (Persaud et al., 1993) is also included in the discussion. The results of the comparison of COPC concentrations detected in the Portneuf River delta sediments to both representative levels and LECs determined whether Phase II would be triggered.

Section 2.7 contains a detailed description of the three triggering criteria. The following briefly describes the three criteria:

- Criterion I was met if an analysis of variance (ANOVA) indicated that a COPC concentration in the Portneuf River delta exceeded a corresponding concentration in a reference (Snake River) delta.
- Criterion II was met if a Student t-test indicated that Portneuf River delta sediment concentrations exceeded Portneuf River sediment concentrations, taken upstream of the EMF facilities as established in Section 4.6.1.1.
- Criterion III was met if the Portneuf River delta sediment concentration exceeded the pre-determined LEC (SII, 1994).

Additional field work, which could have included media sampling, bioassays, and tissue analyses, would have been required if all three of the criteria were met for arsenic, cadmium, mercury, or zinc. Because an LEC was not available for fluoride and selenium, meeting the first two criteria would have resulted in Phase II activities for these COPCs. Based on evaluation of the data using the criteria described above, Phase II was not initiated for any of the COPCs.

Description of Sample Locations

Portneuf River delta sediments were sampled from both the channel and the adjacent mudflats (mudflat samples were more accurately described as floodplain soils when collected above the high water mark). The location within the Portneuf River and Snake River deltas are shown in Table 2.7-3.

Phase I Sediment Analyses

Table 4.6-5 provides a comparison of mean delta COPC concentrations against LECs developed based on lowest AETs observed during EPA Region 10 freshwater bioassay studies, and LELs provided by the OMEE (Persaud et al., 1993). The mean Portneuf River delta sediment COPC concentrations were calculated using the same approach detailed in Section 4.6.1.1 for upstream Portneuf River samples. As discussed previously, EPA has shown interest in using the OMEE guidelines (Persaud et al., 1993) which have therefore been included in the table. However, the LELs are not considered as biologically relevant LECs. As documented in Section 4.6.1.1, these levels are based on questionable techniques and do not adequately differentiate responses of sensitive and insensitive species.

In developing a statistical comparison between Portneuf River delta results and upstream or reference locations, the tendency for COPC sediment concentrations to covary with naturally occurring geochemical parameters was assessed. The degree to which the COPCs tend to bind to sediment particles and thus become less bioavailable is directly related to geochemical properties of the sediment such as pH, redox potential, simultaneously extractable metals (SEM)/acid volatile sulfide (AVS) ratio and content of TOC, aluminum, iron, and clay. In summary, the COPCs cadmium, mercury and zinc are strongly adsorbed to TOC, oxides (primarily aluminum and iron), and clays under alkaline conditions, and tend to form precipitates. The anionic COPCs (fluoride and selenium) may be adsorbed, but to a lesser extent, to some variably charged oxides and organic functional groups. More limited adsorption of these COPCs tends to occur because sediments, like soils, are primarily negatively charged. Finally, the SEM/AVS ratio provides a

measure of the potential binding of cadmium, zinc and mercury (and potentially arsenic and selenium) by sulfides in anoxic sediments.

In order to determine whether correlation exists between sediment COPC concentrations and geochemistry, each sample was also analyzed for the potential covariates listed above. When significant correlation existed between a COPC and a covariate, these data were factored into statistical analyses designed to determine whether COPC concentrations are significantly higher in the Portneuf River delta compared to concentrations in the control (Snake River) delta.

Summary sampling data of the Portneuf and Snake Rivers are provided in Tables 4.6-6 through 4.6-9. The Phase I data were assessed using the triggering criteria previously described, to determine whether Phase II would be needed. Detailed statistical analyses of the results are provided in Appendix S. A summary of the results is presented below.

Arsenic. As a result of a statistical analysis of variance, the Portneuf River delta arsenic concentrations were not found to be significantly higher than Snake River delta concentrations (F-statistic [F]=0.38, degrees of freedom [df]=1,32; probability [p]=0.54). In fact, the Snake River delta sediment concentrations were greater than those of the Portneuf River delta. Consequently, a Phase II analysis was determined to be unnecessary as the first criterion was not met.

Cadmium. The mean concentration in the Portneuf River delta (0.93 mg/kg) was statistically (F=58.44; df=1,32; $p \leq 0.0001$) higher than the mean concentration in the Snake River delta (0.37 mg/kg). This difference persisted after the analysis of covariance was conducted using sediment cadmium concentrations adjusted for levels of the most strongly correlated covariate (aluminum) (F=35.11; df=1,32; $p \leq 0.0001$). Therefore, the first criterion to proceed with Phase II was met.

Less than 50 percent of the upstream Portneuf River sediment sampling data consisted of detected values. Therefore, the second criterion could not be assessed because an insufficient

number of detected upstream Portneuf River sediment samples were available to reliably estimate a mean concentration for statistical analysis.

Consequently, the decision to proceed with Phase II activities depended on a comparison to the LEC. The LEC (7 ppm) is higher than the two detected concentrations in the Portneuf River delta sediments. Consequently, the Portneuf River delta cadmium concentrations, while statistically higher than those detected in the Snake River delta, are unlikely to produce adverse biological effects. This is confirmed by the finding that 10-day bioassays at cadmium levels higher than in the Portneuf River delta (i.e., IWW ditch outfall sediment bioassays) did not show any biological impacts. Therefore, no Phase II assessment of cadmium was warranted.

Fluoride. Fluoride values were significantly higher ($F=18.8$; $df=1,32$; $p\leq 0.0001$) in the Portneuf River delta compared to the Snake River delta. An adequately correlated covariate was not found; consequently, the first criterion to proceed with Phase II was met.

With regard to the second criterion, the Portneuf River delta sediment concentrations were essentially the same as Portneuf River sediment samples collected upstream of the EMF facilities (t-statistic $[t]=-0.37$; $df=24$; $p=0.71$), even after a high upstream outlier value was eliminated. Consequently, a Phase II analysis was determined to be unnecessary for fluoride.

Mercury. A total of 95% of the Portneuf River and Snake River delta sediment mercury samples were non-detects and, therefore, could not be analyzed statistically. Mercury was only detected in two Portneuf River delta mudflat (or floodplain soil) samples, at 0.19 mg/kg (estimated) and 0.46 mg/kg. However, these detected values may be associated with the naturally high variation in regional geological mercury concentrations known to exist in the area (Desborough and Foord, 1992). Furthermore, the highest delta concentration (0.46 mg/kg) was less than the highest concentration detected in Portneuf River sediments located upstream of the EMF facilities (0.55 mg/kg). Therefore, it is unlikely that mercury sediment concentrations in

the Portneuf River delta are significantly higher than levels in either the Snake River delta or upstream of the EMF facilities.

With regard to Criterion III, the two detected Portneuf River delta concentrations are both below the LEC (0.56 mg/kg) based on Region 10 bioassay results. Finally, the sample quantitation limits for the non-detected Portneuf River delta samples were all below the OMEE LEL (0.2 mg/kg). Therefore, Phase II analyses were not warranted for mercury.

Selenium. Because of the skewed nature of the selenium data set, the concentrations were transformed to normalize the distribution. After the data transformation, the differences between the Portneuf and Snake River deltas selenium concentrations were not significant ($F=1.55$; $df=1,32$; $p=0.22$), indicating that Phase II analyses were not necessary for this COPC.

Zinc. Zinc data also required transformation before analysis. The analysis indicated that zinc concentrations were significantly higher in the Portneuf River delta than in the Snake River delta, both before ($F=4.46$; $df=1,32$; $p=0.04$) and after ($F=2.20$; $df=1,31$; $p=0.15$) adjustment for the covariate, iron. Therefore, the first criterion for initiation of Phase II was met. With regard to the second criterion, neither the zinc ($t=-0.15$; $df=25$; $p=0.89$) nor the iron ($t=0.51$; $df=25$; $p=0.61$) concentrations were significantly elevated in the Portneuf River delta, compared to concentrations in upstream Portneuf River sediments. These findings indicate that zinc Portneuf River delta concentrations are equivalent to zinc concentrations upstream of the EMF site, independent of covariate concentrations. Consequently, a Phase II analysis of zinc was determined to be unnecessary.

SEM/AVS Ratio: Indicator of Bioavailability and Toxicity

Several investigators (Ankley et al., 1991; DiToro et al., 1990) have shown that the ratio of simultaneously extractable metals to acid volatile sulfide (SEM/AVS) is a measure of toxicity to *Hyallela* and several other aquatic invertebrates. Specifically, cadmium or (cadmium + nickel) molar SEM/AVS ratios greater than unity may be associated with increased mortality. The

stoichiometry of sulfide salt precipitation in sediments is favored for cadmium and potentially for other divalent cations, at low cation concentrations. At higher concentrations, excess ionic cadmium remains in solution pore-water, resulting in toxicity. Nevertheless, Ankley et al. (1991) demonstrated that the SEM/AVS ratio, rather than the absolute pore-water concentration, was the better predictor of macroinvertebrate toxicity.

AVS is a reliable measure of volatile sulfide only under anoxic conditions (Allen et al., 1993). Under oxic conditions, endogenous ferrous sulfide is oxidized to ferric sulfate, invalidating the use of the SEM/AVS ratio. Accordingly, most of the AVS measurements (including virtually all of the well oxygenated mudflat samples) in both delta systems were non-detects. Furthermore, the absolute SEM value for a specific cation is not a good measure of toxicity (Ankley et al., 1991). Consequently, neither SEM/AVS nor the individual SEM values were able to be used to assess the potential bioavailability or toxicity of COPCs in Portneuf River delta sediments.

Summary of the Delta Investigation

The levels of arsenic and selenium in Portneuf River delta sediments were either less than or essentially equivalent to the Snake River delta sediments. With respect to fluoride and zinc levels, concentrations within sediments collected from the Portneuf River upstream of the EMF facilities were either statistically similar or greater than those in the Portneuf River delta. Although Portneuf River delta mercury levels could not be analyzed statistically against upstream levels, the two detected delta concentrations were both lower than the one mercury concentration detected in Portneuf River sediment sampled from upstream of the EMF facilities. In addition, the two detected delta concentrations were below the LEC, and the sample quantitation limits for the remaining non-detected Portneuf River delta samples were all below the more conservative OMEE LEL.

Only the cadmium analysis relied completely upon a comparison to an LEC to determine the need to proceed with a Phase II assessment. The detected cadmium levels were all lower than the

LEC based on the lowest AET observed during Region 10 freshwater bioassay studies and were lower than the highest value obtained during the IWW ditch outfall study (Section 4.6.1). No effects on benthic biota were observed during the toxicity tests conducted as part of the IWW ditch outfall study, suggesting that the lower delta cadmium concentration would also have no biological effect.

As a result of the analyses detailed in the section, and described in greater detail in Appendix S, none of the COPCs of the Phase I delta sediment analyses met all three triggering criteria. Therefore, Phase II of the Portneuf River Delta Study was determined to be unwarranted and was not implemented since no EMF related impacts were apparent in the Portneuf River delta.

4.6.2 TERRESTRIAL INVESTIGATION

The terrestrial investigation was performed according to the ESAP, with some modifications. The methods, and modifications, are summarized in Section 2.7. The objectives of the terrestrial investigation were to collect data that would allow comparison of COPC concentrations to concentrations at reference (background) locations, and to provide site-specific estimates of bioavailability relevant to both human and ecological receptors. Bioavailability was assessed through both soil mineralogical analyses and the analytical determination of COPC concentrations in ecological receptors (i.e., vegetation and small mammals), because these data provide an empirical measure of the transport and biological uptake of COPCs from soils. Although this section does not quantitatively address the potential ecological impacts associated with COPC concentrations in ecological receptors, available information on levels of concern are referenced where appropriate.

In the following, the results of the field investigation on terrestrial habitats within the EMF study area are discussed. The investigation was performed on the two predominant EMF terrestrial ecosystems: 1) the sagebrush steppe habitat, the dominant native upland terrestrial ecosystem; and 2) the riparian habitat bordering the Portneuf River. Habitat descriptions and potential

ecological receptors were previously provided in Section 3.7. In the sagebrush steppe habitat, soils, vegetation, and small mammals were collected and analyzed for cadmium, fluoride and zinc; the three constituents selected as COPCs by EPA in this investigation. Vegetation and soils (but not small mammals) were collected from the riparian habitat bordering the Portneuf River for COPC analyses.

The data are presented separately for each sampled media (soils, vegetation and small mammals). For each media, a comparison of the COPC concentrations detected in potentially impacted areas to concentrations at the reference location are initially presented. A discussion of the potential significance of the data with respect to food chain transfer (e.g., plant uptake and subsequent transfer through higher orders of the food chain) is then presented. The method of data treatment for determining summary COPC concentration statistics for use in these analyses was the same as previously described for the aquatic investigation (Section 4.6.1.1), with the exception that mean concentrations were calculated regardless of the percent of non-detect results for a given analyte. As before, one-half of the sample quantitation limit was used in the calculations for all non-detect results.

4.6.2.1 Soil Analyses

Soils were obtained from 10 subplots at each of the five terrestrial sampling locations, including the reference locations. Each sample was then analyzed for the three COPCs; cadmium, fluoride and zinc. For the two terrestrial habitats assessed, the following discussion provides: 1) a comparison of COPC concentrations in potentially impacted soils to levels at the reference location; and 2) the findings of soil mineralogical studies designed to determine the mineralogical form of COPCs within both sagebrush steppe and riparian soils. As previously detailed, the mineralogical analyses were performed to aid in the assessment of COPC bioavailability from EMF soils.

Soil Concentration Comparisons: Reference versus Potentially Impacted Areas

Sagebrush Steppe Habitat. A comparison of the mean (\pm standard deviation) and median soil sample concentrations for the three COPCs at each of the three sampling locations is provided in Table 4.6-10. Appropriate summary statistics for other chemical and physical soil parameters (calcium, TOC, pH, and cation exchange capacity) are also provided. These parameters are used to define differences in soil chemistry between sampling locations.

Soil concentrations of all three COPCs are elevated in the two potentially impacted locations on the Michaud Flats and Bannock Hills, relative to the reference (Ferry Butte) location. As presented in Table 4.6-10, cadmium levels were elevated above the reference concentration to a greater extent than both fluoride and zinc.

The bioavailability of COPCs is directly equivalent to the exposure of ecological receptors (e.g., vegetation). Therefore, bioavailability is an important determinant in assessing the potential for the elevated COPC soil concentrations to impact these receptors. As discussed in Section 3.4, due to the high pH and calcareous nature of potentially impacted soils (and potentially high TOC), cadmium and zinc are probably present as insoluble precipitates and/or adsorbed to soil particles. Thus, COPC soil pore-water concentrations, which constitute the bioavailable fraction of COPCs within the soil matrix, will tend to be low suggesting limited availability for plant uptake. Vegetation and plant uptake analyses, discussed in Section 4.6.2.2, confirm this hypothesis.

Riparian Habitat. Table 4.6-11 presents soil concentrations of the three COPCs, and other soil parameters, in the potentially impacted (Portneuf River) and reference (Snake River) riparian habitat sampling locations. Concentrations of all three COPCs at the Portneuf location are moderately to markedly elevated above the Snake River concentrations. Three other measured soil parameters (calcium, TOC, and cation exchange capacity), that are often associated with increased divalent cation adsorption, are also elevated in the Portneuf River riparian habitat and may therefore partially account for the elevated zinc and cadmium COPC

concentrations observed in this area. However, these parameters are not sufficiently elevated to fully explain the higher Portneuf River habitat COPC concentrations. Therefore, it is likely that the increased Portneuf COPC concentrations, relative to the reference location, are also attributable to EMF-related activities. However, due to the elevated levels of calcium, TOC and cation exchange capacity, it is again likely that soil pore-water concentrations and, hence, bioavailability will be low.

Mineralogical Analyses

A knowledge of the predominant mineralogical forms of a COPC in soil can result in the development of geochemical models useful for predicting COPC pore-water concentrations, which are the best estimates of bioavailability to ecological receptors. As discussed in Section 2.7, mineralogical analyses were to be performed as part of the ecological investigation to determine the mineral forms and oxidation states of fluoride, cadmium, zinc, and arsenic in both soils at each of the five terrestrial sampling locations, and source materials present at the EMF facilities. (Arsenic was added to the COPC list because of potential significance to human health risk.)

The results of a feasibility study conducted on the most highly impacted terrestrial soil sample (Bannock Hills sagebrush steppe habitat) and concentrated on-site source material (slag at FMC) indicated that arsenic, cadmium, and zinc concentrations were below the detection limits for speciation analysis. In addition, fluoride speciation could not be performed because fluoride has a low molecular weight and was subject to background interference that is associated with most phyllosilicates. In an attempt to concentrate the mineralogic phases of the metal COPCs, the soil and source material samples were treated using a heavy liquid separation technique. However, the resulting concentrations were still too low to perform statistically reliable speciation. The concentrations of the metal COPCs detected in five separate particle size fractions of the Bannock Hills soil sample are presented in Table 4.6-12.

Although no specific mineralogical data were derived, a qualitative evaluation of soil COPC bioavailability can be made using known chemical and physical soil parameters. Due to the alkaline and calcareous nature of the EMF soils, arsenic may exist primarily as calcium arsenate or other poorly soluble forms (Sadiq, 1981). Similarly, fluoride may be present primarily as fluorapatite or calcium and aluminum fluorides (Kabata-Pendias and Pendias, 1992). These minerals all exhibit very low solubilities and, consequently, low bioavailability. Geochemically, cadmium behaves similarly to zinc. In the soil, both cadmium and zinc may exist as carbonates, phosphates, and other poorly soluble salts (Bohm et al., 1985). In general, the solubilities of these minerals is reduced two orders of magnitude with each unit pH increase (Lindsay, 1979).

Thus, the geochemistry of soils in the EMF study area indicates that the bioavailability of the COPCs is likely to be very limited. Because the mineralogical analyses were not feasible, this conclusion is best tested based on the ratio of plant tissue-to-soil concentrations at each of the sampling locations. This is discussed in the next section.

4.6.2.2 Vegetation Analyses

At the sagebrush steppe habitat locations, 20 subplots were sampled for sagebrush and 10 subplots were sampled for wheatgrass. Only Russian Olive fruits were sampled in the riparian habitats; 10 subplots were sampled at each riparian habitat location. The following discussion provides: 1) a comparison of vegetation concentrations in potentially impacted areas to reference locations; 2) an evaluation of bioavailability as measured by the ratio of plant-to-soil COPC concentrations (i.e., plant uptake); and 3) a comparison of vegetation concentrations to levels which may potentially impact ecological receptors.

Vegetation Concentration Comparisons: Reference versus Potentially Impacted Areas

Sagebrush Steppe Habitat. Table 4.6-13 contains the mean (\pm standard deviation) and median vegetation sample concentrations of the COPCs at each of the three sagebrush steppe habitat sampling sites. A comparison of washed versus unwashed concentrations is also detailed

in the table to provide information about the extent of current airborne deposition of COPCs onto vegetation. Because of the presence of fluoride in the detergent used to wash the vegetation (Section 4.1), only the cadmium and zinc analyses of washed vegetation were considered appropriate. It should also be noted that the grasses were dormant and dehydrated when sampled and, therefore, represent the highest concentrations as discussed in Section 2.7.

As shown in Table 4.6-13, the differences between potentially impacted and reference area zinc concentrations are negligible. These results indicate that vegetation viability, and the resultant impacts to herbivores through the food chain, are not a concern for zinc in this habitat. By contrast, both potentially impacted areas were found to contain elevated fluoride levels in sagebrush, and elevated cadmium concentrations in both sagebrush and grasses. Fluoride concentrations in grasses were only elevated in the Bannock Hills sampling location relative to the reference (Ferry Butte) location. During the field investigation, no direct impacts on vegetation growth or viability were observed.

A comparison of washed versus unwashed sagebrush samples for cadmium and zinc suggests that unwashed samples contain between 7 to 23 percent higher COPC concentrations than washed samples (Table 4.6-13). Therefore, foliar deposition from current airborne sources appears to contribute only a small proportion of the total vegetation COPC concentrations (i.e., plant uptake of COPCs from soils is the predominant process).

Riparian Habitat. Despite the elevated COPC concentrations in potentially impacted soils of the Portneuf River riparian habitat, Russian Olive concentrations of cadmium and fluoride were below detection limits, comparable to the Snake River riparian habitat location (Table 4.6-15). Furthermore, the Portneuf riparian zinc concentrations in the Russian Olives were only slightly elevated over the reference concentrations.

The low levels observed in the Russian Olives demonstrate that a representative fruit in the potentially impacted riparian habitat was not adversely affected despite elevated soil

concentrations at the sampling location. As previously detailed, this finding is likely associated with the limited root uptake of COPCs from the alkaline (pH 7.2 to 8.6) and calcareous soils typical of both riparian and sagebrush steppe soils in the EMF study area. The limited uptake of COPCs in both habitats is further demonstrated by the calculation of uptake parameters, discussed in the next section.

Plant Uptake

Plant uptake of chemicals is a complex function of many parameters including plant species, soil type, cultivation methods, climate, length of growing season, etc. However, for a given plant species, the soil conditions are a critical factor governing uptake.

It has been well-established that plant roots take nutrients and chemical constituents primarily from soil pore-water or soil solution. Dissolved soil pore-water constituents are usually a small fraction of the total chemical concentration in soils and depend strongly on chemical, biological, and physical properties and processes. Of all soil parameters, soil pH, often known as the master parameter, is the single most important factor controlling soil pore-water chemical concentrations (metals in particular) and is therefore an important determinant of plant uptake. In general, metals tend to form precipitates and/or absorb to soil particles and organic matter at high pH. For cadmium and zinc, each unit increase in pH can result in a 100-fold decrease in solubility of cadmium or zinc bearing minerals (Lindsay, 1979), greatly reducing their bioavailability to ecological receptors.

The soils in the EMF area are typical of those in the northwestern United States (Foth and Turk, 1972; SCS, 1977). Due to the alkaline pH and calcarious nature of the soils, metals are primarily present as precipitates and/or adsorbed forms. Consequently, pore-water metal concentrations are very low and, hence, plant uptake is expected to be low. This is supported by a review of site-specific plant uptake parameters, which were derived from the ratio of plant-to-soil concentrations (i.e., the ratio of vegetable (or fruit) tissue concentration to the total soil

concentration), obtained from the ecological investigation. For comparison purposes, the site-specific uptake findings have been compared to default uptake estimates provided by Baes et al. (1984).

Baes et al. (1984) derived plant-to-soil concentration ratios for inorganic chemicals by reviewing available literature detailing chemical concentrations in a variety of agricultural crops and soils. The Baes default ratios shown in Tables 4.6-15 and 4.6-16 were estimated from various published reports in which either vegetation concentrations, or the uptake parameter, often ranged over several orders of magnitude for a specific chemical. The default ratios cited by Baes et al. (1984) were simply calculated by averaging the available plant-to-soil concentration ratio estimates for a chemical. When the vegetation concentrations were reported without a corresponding soil concentration, a geologic mean soil concentration was adopted. In such instances, a plant tissue concentration was obtained by averaging available literature values based on a variety of plants. For nearly all elements, only a limited number of samples were available to calculate the average plant uptake ratio, and sometimes the calculation relied on a single reference. In summary, the Baes et al. (1984) default uptake parameters are not considered suitable for characterizing plant uptake in the EMF area. In fact, in recognition of the great variability in soil conditions and plant uptake, Baes et al. (1984) suggest that site-specific data be used to characterize uptake coefficients when available.

With respect to the EMF site, a comparison of plant uptake in the reference locations to the Baes default parameters demonstrates that uptake in the natural soils is generally less than the average default parameter. This finding is expected due to the high binding effect of the alkaline and calcareous EMF area soils. However, it is also apparent in Tables 4.6-15 and 4.6-16 that the uptake ratio is significantly lower in potentially impacted areas than both the Baes default parameters and the parameters calculated for the reference locations. The low plant uptake in the potentially impacted plots is most likely associated with poor dissolution and, hence, low bioavailability of inorganics deposited onto soils through EMF-related activities. These

constituents appear to primarily precipitate and/or adsorb to soil particles, making them less available for plant uptake.

Levels of Ecological Concern

In the EMF study area, mule deer are a representative ruminant which may be exposed to COPCs via the lush riparian vegetation in the late summer and fall, and via big sagebrush and related plants in winter and early spring (Ingles, 1965; Taylor, 1956; Thomas, 1991; Wallmo, 1981; Welch, 1983). However, exposure to sagebrush is largely dependent on the severity of the winter. Mule deer will move into areas supporting big sagebrush during severe winters characterized by heavy snow accumulation.

The following discussion provides a preliminary assessment of potential COPC-related impacts to mule deer by comparison to levels of concern derived from the available literature.

Cadmium. The kidney is the principal target organ of cadmium toxicity. The most relevant study of the significance of slightly elevated vegetation cadmium concentrations to grazing ruminants is a study by Chaney et al. (1988). This study investigated cadmium tissue distribution and toxic effects in ruminants exposed to zinc smelter emissions in Palmerton, Pennsylvania. At the most heavily contaminated site, dry matter vegetation concentrations ranged from 10.3 to 12.1 mg/kg. The kidney of a cow slaughtered from this site contained 87.2 mg cadmium/kg dry weight (between 9 and 29 mg cadmium/kg wet weight (EPA, 1993c) which is below a level associated with significant proteinuria or morphologic changes in the kidneys.

Compared to levels in the EMF-impacted areas, the Palmerton vegetation concentrations, were approximately an order of magnitude higher than the unwashed sagebrush concentrations. Because the higher vegetation concentrations at Palmerton did not result in toxic cadmium concentrations in the kidney, the primary target site, the dietary intake of vegetation by mule deer, and other ruminants, in the EMF study area is unlikely to result in any significant cadmium-related health risk.

Fluoride. The clinical signs of fluoride intoxication are osteo (skeletal) and dental fluorosis (disruption of the bone and tooth structure, respectively). These effects are particularly severe in ruminants (mammals with multiple stomachs, such as cattle and deer).

Shupe et al. (1963) provided evidence that the long-term feeding of 27 mg/kg fluoride to dairy cattle, in (dry weight) fodder, was associated with minor dental fluorosis and slight skeletal changes. Adverse osteofluorosis and dental fluorosis clearly occurred at the next highest concentration, 49 mg/kg. These findings receive support from other continuous feeding studies (Newell and Schmidt, 1958; Suttie et al., 1985) in cattle and white-tailed deer. In a review of experiments involving over 200 (primarily field) cattle, Shupe et al. (1992) determined three distinct exposure groups. Moderate to severe dental and osteofluorosis occurred in cattle exposed to 60 to 93 mg/kg of fluoride in vegetation, mild to moderate fluorosis appeared in cattle exposed to 25 to 60 mg/kg, and minimal dental changes occurred in cattle exposed to vegetation containing less than 25 mg/kg fluoride.

At the Bannock Hills and Michaud Flats sampling locations, the mean unwashed sagebrush fluoride concentrations were 74 and 52 mg/kg (dry weight), respectively. According to the dairy cattle and white-tailed deer data discussed above, long-term ingestion of these concentrations may result in fluorosis. However, the results of industrial exposure studies in mule deer and pronghorn antelope suggest that somewhat higher vegetation fluoride intake levels are needed for the expression of fluorosis (Kay et al., 1975; Newman and Murphy, 1979; Woodward-Clyde, 1993).

The Woodward-Clyde (1993) study is a risk assessment of potential fluoride toxicity in pronghorn antelope exposed near a Wyoming phosphates facility. In general, forage, which was analyzed monthly for over two years, had mean fluoride concentrations slightly to moderately higher than concentrations measured at impacted EMF locations. Using a simulation of daily antelope activity, the Woodward-Clyde investigators concluded that individual antelopes could not realistically consume enough contaminated forage to develop fluorosis. This conclusion was

supported by measured bone fluoride concentrations that were generally less than 1500 mg/kg (dry weight) in 2.5 to 8.5-year old antelope, significantly less than the 2400 mg/kg (4000 mg/kg ash weight) at which clinical signs of toxicity (except for minor tooth abrasions) may begin to appear (Walton, 1988; Woodward-Clyde, 1993). Furthermore, tooth erosion was either negligible or not apparent in all animals. Based exclusively on these data, mule deer chronically exposed in the EMF study area should not suffer adverse fluoride-related effects.

Zinc. As indicated in Table 4.6-13, zinc concentrations in unwashed sagebrush were essentially identical at the potentially impacted and reference locations. Because of the similarity of zinc vegetation levels and the finding that zinc toxicity only occurs at very high intake levels (Andrews et al., 1989; Chaney et al., 1988; Talmadge and Walton, 1990), mule deer should not be considered at risk from zinc intoxication in the EMF study area.

4.6.2.3 Small Mammal Analyses

Small mammals were sampled only in the sagebrush steppe habitats. Deer mice were the target species, however, harvest mice were also collected in some areas (Section 2.7). Ten mice were collected for analysis of COPCs from each of the two potentially impacted sagebrush habitat locations (Bannock Hills and Michaud Flats) and the reference sagebrush habitat location (Ferry Butte). Between 4 and 6 mice of each sex were sampled at each location.

The following discussion provides: 1) a comparison of small mammal concentrations of cadmium, zinc, and fluoride (whole body and fluoride levels in the femur) in potentially impacted areas to reference locations; and 2) a comparison of small mammal concentrations of cadmium, zinc, and fluoride to levels which may potentially impact ecological receptors.

Small Mammal Concentration Comparisons: Reference versus Potentially Impacted Areas

Summary statistics for whole body and femur (fluoride) concentrations of cadmium, zinc, and fluoride in mice, harvested at the three sampling locations, are provided in Table 4.6-18. In

general, mouse cadmium and fluoride concentrations at the potentially impacted locations showed statistically significant increases ($p < 0.05$) compared to the concentrations measured at the reference (Ferry Butte) location. The concentrations of fluoride in femurs of mice on the EMF site were also lower than concentrations of fluoride in femurs of voles fed fluoride in diets prepared from grass grown on contaminated sites (Boulton et al. 1994). In the Boulton et al. (1994) study, only slight dental changes were observed in voles fed a high fluoride diet for 84 days resulting in femur concentrations of 909 mg/kg dry weight, while decreased body weight gain, marked dental lesions and early mortality was observed in voles with femur fluoride concentrations of about 2100 mg/kg dry weight.

As seen in Table 4.6-17, mice collected from Bannock Hills had, on the average, threefold higher ($p < 0.001$) whole body cadmium concentrations than mice collected from Michaud Flats. Average whole body fluoride concentrations were also higher (1.4 fold greater; $p < 0.05$) at the Bannock Hills location although the average femur concentrations were higher in mice harvested at Michaud Flats (633.3 ± 220.3 mg/kg versus 290.8 ± 197.6 mg/kg at Bannock Hills; $p < 0.01$). The higher femur concentrations of fluoride in mice from Michaud Flats location may be attributable to older mice being trapped at Michaud Flats relative to Bannock Hills since fluoride concentrations in bone and would be expected to be higher given a longer exposure period. The average mouse weight at Bannock Hills (13 grams) suggests that the trapped mice were young-of-the-year, and that the measured fluoride levels were not representative of lifetime exposure. By contrast, the average mouse weight at Michaud Flats and Ferry Butte (17.1 and 17.5 grams, respectively), indicate that the mice were more mature, and suggest that the observed femur concentrations reflect a longer-term exposure.

Only small amounts of fluoride in bone are bioavailable to other animals that ingest bone. In general, 99 percent of whole body fluoride is associated with calcified tissue, i.e. bone (NRC, 1993). To estimate the percentage of whole body fluoride that is found in the skeleton, based on the femur concentration, would require the calculation of mouse body weight comprised of

calcified tissues which is likely to vary with age of the mice. Therefore, one estimate applied to all the mice collected at the site would not be accurate. The use of femur fluoride to represent fluoride in the rest of the skeleton may also underestimate the concentration of fluoride in bone. A study in white-tailed deer (Suttie et al., 1985) in which the deer were fed fluoride for about 2 years reported that the mandible contained a higher concentration of fluoride than the femur. The difference in concentration of fluoride between the two bones decreased with increasing fluoride exposure. Therefore, if the mandible or other bones of mice contained more fluoride than the femur, the amount of fluoride associated with the skeleton would be underestimated, unless fluoride exposure was sufficient to reduce the differences in concentrations between the femur and mandible.

It is also important to note that mice were not washed before fluoride analysis, so that total body fluoride measurements also include fluoride in soil on the fur of animals which may contribute to the variability in the mouse fluoride concentration between Michaud Flats and Bannock Hills. As smaller animals have a greater surface area relative to body weight, more of the total body fluoride in the smaller animals may be associated with soil in their fur, and may therefore result in an underestimate of the amount of fluoride found in the skeleton of smaller animals. Fluoride associated with soil in the fur could also account for some of the variability in whole body fluoride. Fluoride associated with fur may not be ingested by predators depending on feeding habits. Some predators pull apart prey (e.g., fox), while others swallow everything (e.g., raptors).

Zinc concentrations in mice did not differ between the impacted and reference locations. Consequently, no further consideration of zinc levels in mice are considered in the remaining discussion, since no impact is apparent.

Levels of Ecological Concern-Small Mammals

Cadmium. Kidney tissue concentrations of cadmium were not available from this analysis. Nevertheless, mouse whole body data suggest that cadmium tissue concentrations are well below levels expected to result in proteinuria (abnormal leakage of blood proteins into the urine occur are kidney concentrations of 200 mg/kg) or related kidney effects (Nordberg et al., 1992).

Fluoride. A fluoride skeletal tissue concentration of ecological concern for deer mice can be estimated by analogy to results in livestock which have been studied more extensively, as discussed previously. The fresh weight of the mice femurs obtained from the potentially impacted areas were significantly lower than bone fluoride concentrations found by Walton et al. (1988) to result in clinical signs of toxicity in livestock. Walton (1988), Shupe et al. (1992), and Newman and Markey (1976) also noted that rodents are more tolerant of fluoride intoxication than cattle. The concentrations of fluoride in femurs of mice in the vicinity of the EMF facilities were also lower than concentrations of fluoride in femurs of voles fed fluoride in diets prepared from grass grown on contaminated sites (Boulton et al., 1994). In the Boulton et al. (1994) study, only slight dental changes were observed in voles fed a high fluoride diet for 84 days resulting in femur concentrations of 909 mg/kg dry weight, while decreased body weight gain, marked dental lesions and early mortality was observed in voles with femur fluoride concentrations of about 2100 mg/kg dry weight. Consequently, the average femur concentrations at Michaud Flats are below levels expected to be associated with significant signs of skeletal damage in mice. Although interpretation of the Bannock Hills results is confounded by the young age of the mice, the available bone concentrations suggest that mice in the vicinity of this location are also not significantly impacted by fluoride.

Levels of Ecological Concern - Top Predators:

The predators which prey on small mammals in the sagebrush steppe habitat are discussed in Section 3.7. Representative species of these top predators are the barn owl, red-tailed hawk and

fox. All three predator species have a home range that is considerably larger than the home range of their typical prey, such as the deer mouse (Stickel, 1968). Recent studies (Thomson, 1987) have suggested that predators are equally likely to choose prey in a moderately contaminated area as in a non-impacted area. Consequently, daily COPC intake of predators will likely be related to average prey tissue concentrations of prey that include multiple prey communities with different exposure histories. More details of the feeding habits and local ecology of the representative top predators are provided in Section 3.7. The following provides an overview of the potential impacts of small mammal COPC concentrations on these top predators based on levels of ecological concern established through toxicological studies.

Cadmium. For the fox and the red-tailed hawk, much of the prime hunting habitat is likely to be north of the Michaud Flats sampling location. In these areas, cadmium concentrations of mice will generally be no greater, and often less, than the tissue concentrations found at Michaud Flats. The barn owl, which is more accustomed to anthropogenic structures, may be more likely to hunt nearer to Michaud Flats. Nevertheless, a significant portion of this species' catch may also be derived from agricultural fields, which are more distant from the Michaud Flats sampling area.

Each of the individual predators consumes the equivalent of approximately four (juvenile owls) to twenty (adult foxes) mice daily (Craighead and Craighead, 1956; Sargeant, 1978; Taylor, 1994). In general, cadmium does not produce adverse effects in mammals and birds at oral doses less than 1 mg/(kg x day) (Anwar et al., 1961; Cain et al., 1983; Fingerle et al., 1982; Richardson et al., 1974). Based on the mouse tissue concentration results, even in the improbable event that a sufficient number of prey could be taken exclusively from the Bannock Hills location, the total cadmium intake would not approach the 1.0 mg/(kg x day) level necessary for adverse effects. Consequently, cadmium does not appear to pose a toxicological hazard to any of the top predators.

Fluoride. Numerous investigators have studied the health effects of soluble fluoride in birds and mammals. In screech owls, long-term daily feeding of 56.5 mg/kg fluoride (as sodium fluoride) resulted in smaller egg volumes (Hoffman et al., 1985; Pattee et al., 1988). Slight but more significant reproductive effects, including smaller egg weight and a shorter leg bone length (compared to controls) occurred at 232 mg/kg. In mammals, skeletal and reproductive effects in minks and foxes are found after long-term exposure to more than 100 mg/kg fluoride wet weight (Aulerich et al., 1987; Eckerlin et al., 1986; Shupe et al., 1987). However, these long-term feeding studies using soluble fluoride are not directly comparable to uptake of fluoride from the whole mouse.

Fluoride preferentially deposits in skeletal tissue and therefore is not likely to be bioavailable. Furthermore, predators preferentially ingest soft tissue or, if the whole body is ingested, the skeleton may not be digested. For example, barn owls consume little of their prey's skeleton (Thomson, 1987) and their relatively alkaline stomach pH (Smith and Richmond, 1972) is largely incapable of leaching fluoride from bone. Therefore, for comparison between the health effects of fluoride in top predators based on soluble fluoride and health effects of ingestion of fluoride in small mammals, the bioavailable fraction of fluoride was assessed by subtracting the skeletal concentrations from the whole body concentrations. In this exercise, the femur fluoride concentrations in Table 4.6-18 were conservatively considered representative of total skeletal concentrations. Considering EMF mouse tissue concentration results (Table 4.6-17) and adjusting for the skeletal tissue concentrations, the literature data suggest that barn owls are unlikely to be susceptible to fluoride exposure in the EMF study area.

Birds of prey that hunt by day, such as the hawk, tend to accumulate fluoride more readily than nocturnal hunters, such as the owl (Seel and Thomson, 1984). However, because of the hawk's large home range (U.S. Department of the Interior, 1979), intake of impacted food sources will likely constitute only a minor proportion of this species' average daily intake. In summary, it is

unlikely that birds of prey within the EMF study area will be susceptible to fluoride exposure in the EMF study area.

With respect to foxes, chronic exposure to small mammals residing in the potentially impacted areas is not likely to result in an adverse effect, particularly since the preferential hunting habitat for foxes should be the relatively non-impacted agricultural fields. Furthermore, in support of the assumption that foxes may not digest the entire skeleton of their prey, Walton (1988) cited studies in which foxes had lower bone fluoride concentrations than the voles and mice they were eating, despite the fox's longer life span.

The results of these analyses provide evidence that EMF-related fluoride exposure is not likely to result in a health risk to representative top predators.

4.6.2.4 *Summary and Conclusions of the Terrestrial Assessment*

The data obtained from the terrestrial investigation indicate that EMF-related impacts are apparent in soils near the facilities. However, vegetation and small mammal analyses indicate that there is not a linear increase in cadmium, fluoride and zinc concentrations in ecological receptors when compared to nearby soil levels. Furthermore, under realistic exposure conditions, EMF-related cadmium, fluoride, and zinc small mammal concentrations are unlikely to result in adverse health effects to top predators. Vegetation concentrations of cadmium and zinc will have no adverse effects to ruminants that may graze in the EMF study area. Because of the mule deer activity patterns, extended mule deer grazing at contaminated locations is unlikely. Consequently, fluoride exposures will not be sufficient to result in adverse health effects in these species.

Surface and Subsurface Characterizations
Tables for Section 4.6

TABLE 4.6-1
RESULTS OF THE IWW DITCH OUTFALL SEDIMENT AND SURFACE WATER SAMPLING, CONDUCTED
SEPTEMBER, 1994

Parameters (mg/kg)	IWW Ditch Outfall Sediment Stations		
	SD16 (downstream of outfall)	SD17 (at outfall)	SD21 (upstream of outfall)
Aluminum, total	3480	1710	1930
Antimony, total	0.80 J	0.64 J	0.67 J
Arsenic, total	2.4	2.5	2.1
Barium, total	75.8	102	49.1
Beryllium, total	0.21	0.15	0.11
Boron, total	10.3 U	6.6 U	8.0 U
Cadmium, total	0.7	3.2 J	0.44 J
Calcium	104,000	72,500	95,600
Chromium, total	10.6 J	7.0 J	5.1 J
Cobalt, total	1.7 U	2.7	2.4
Copper, total	4.9	12.9	1.3
Fluoride, total	245 J	312 J	183 J
Iron	6990	4850	7290
Lead, total	5.3 J	3.9 J	6.4 J
Magnesium	3120 J	1640 J	2430 J

TABLE 4.6-1 (continued)
RESULTS OF THE IWW DITCH OUTFALL SEDIMENT AND SURFACE WATER SAMPLING, CONDUCTED
SEPTEMBER, 1994

Parameters (mg/kg)	IWW Ditch Outfall Sediment Stations		
	SD16 (downstream of outfall)	SD17 (at outfall)	SD21 (upstream of outfall)
Manganese, total	214 J	1120 J	190 J
Mercury, total	0.05 U	0.05 U	0.06 U
Molybdenum, total	34.1	22.8	28.4
Nickel, total	6.0	6.6	5.4
Potassium	886	587	358
Selenium, total	0.36 UJ	0.29 UJ	0.28 UJ
Silver, total	2.3 J	2.1 J	2.1
Sodium	135	224	80
Thallium, total	0.37 U	0.30 U	0.26 U
Vanadium, total	10.2 J	6.4 J	4.2 J
Zinc, total	27.7	49	18.3
Ammonia	15 J	3.4 J	7.8 J
pH	8.48	8.46	8.38
TOC	14600 J	18000 J	20300 J
AVS	0.19	0.18	0.20

TABLE 4.6-1 (continued)
RESULTS OF THE IWW DITCH OUTFALL SEDIMENT AND SURFACE WATER SAMPLING, CONDUCTED
SEPTEMBER, 1994

Parameters (mg/kg)	IWW Ditch Outfall Sediment Stations		
	SD16 (downstream of outfall)	SD17 (at outfall)	SD21 (upstream of outfall)
SEM Cadmium ($\mu\text{mole/g}$)	0.00062 J	0.0019 J	0.00017 UJ
SEM Copper ($\mu\text{mole/g}$)	R	R	R
SEM Lead ($\mu\text{mole/g}$)	0.012 J	0.0060 J	0.0081
SEM Nickel ($\mu\text{mole/g}$)	0.041 J	0.018 UJ	0.082 J
SEM Zinc ($\mu\text{mole/g}$)	0.063 J	0.080 J	0.029 UJ
Conductivity ^(a) (mS/cm)	0.623	0.608	0.573
Turbidity (NTU) ^(a)	7.67	8.0	11.3
Dissolved O ₂ ^(a) (mg/L)	7.88	8.90	11.15
Temperature ^(a) (°C)	13.8	13.6	14.3
Salinity ^(a) (0/00)	0.02	0.02	0.02
Flow ^(a) (ft/sec)	2.17	1.68	0.46
Depth ^(a) (inches)	10	10.3	15.3

Notes: J = Estimated value.

U = Not detected/Below detection limit; sample quantitation limit is provided.

R = Rejected value.

^(a) = These physical parameters are average values of three samples collected at each station.

TABLE 4.6-2
COMPARISON OF CONCENTRATIONS OF COPCS AND OTHER TOTAL METALS IN IWW DITCH OUTFALL SEDIMENTS TO UPSTREAM PORTNEUF RIVER
SEDIMENTS AND REPRESENTATIVE SOIL LEVELS

Parameter (mg/kg)	IWW Ditch Outfall Sediment Stations			Upstream Stations							Mean Upgradient Concentration	Representative Soil Level
	SD16	SD17	SD21	SD21	SD22	SD23	SD24	SD25	SDA1	SDA2		
Aluminum, total	3480	1710	1930	4450	5230	11400	10400	4850	14600	11200	8886	13900
Arsenic, total	2.4	2.5	2.1	3.6	3.4	3.5	4.1	4.1	5.7	5.5	4.3	7.7
Cadmium, total	0.7	3.2 J	0.44 J	0.82	0.29 U	0.36 U	0.33 U	0.16 U	0.43 U	0.45 U	NC	1.9
Fluoride, total	245 J	312 J	183 J	198	500	R	241	193	460	390	330	600
Iron, total	6990	4850	7290	7850	11400	10400	7730	7730	14400 J	10900 J	10080	14400
Mercury, total	0.025 U	0.025 U	0.03 U	0.11 U	0.08 U	0.14 U	0.19 U	0.08 U	0.55	0.19 U	NC	0.16
Selenium, total	0.18 U	0.145 U	0.14 U	0.17 U	0.19 U	0.21 U	0.54	0.72	0.75 U	0.65 U	NC	3.05
Zinc, total	27.7	49	18.3	40.1	32.7	48.1	42.9	24.3	55.3 J	50.3 J	42	52.8

Notes: Units in mg/kg.

J = Estimated Value.

U = Not detected/below detection limit.

R = Rejected value; high outlier.

NC = Half or more of the sample concentrations were non-detected, and a mean concentration was therefore not calculated for use in the analysis.

TABLE 4.6-3
COMPARISON OF CONCENTRATIONS OF COPCS AND COVARIATES IN IWW DITCH OUTFALL
TO LEVELS OF ECOLOGICAL CONCERN

Parameter (mg/kg)	IWW Ditch Outfall Sediment Stations			LEC ^(a)	OMEE - LEL ^(b)
	SD16 (downstream of outfall)	SD17 (at outfall)	SD21 (upstream of outfall)		
Aluminum, total	3480	1710	1930	-	-
Arsenic, total	2.4	2.5	2.1	40	6
Cadmium, total	0.7	3.2 J	0.44 J	7	0.6
Fluoride, total	245 J	312 J	183 J	-	-
Iron, total	6990	4850	7290	-	-
Mercury, total	0.025 U	0.025 U	0.03 U	0.56	0.2
Selenium, total	0.18 U	0.145 U	0.14 U	-	-
Zinc, total	27.7	49	18.3	490	120
AVS Sulfide	0.19	0.18	0.20	-	-
SEM Cadmium	0.00062 J	0.0019 J	0.00017 J	-	-
SEM Copper ^(c)	0.0026 J	0.0025 J	0.014 UJ	-	-
SEM Lead ^(c)	0.012 J	0.0060 J	0.0081 J	-	-
SEM Nickel ^(c)	0.041 J	0.018 UJ	0.082 J	-	-
SEM Zinc ^(c)	0.063 J	0.080 J	0.029 UJ	-	-
TOC	15000 J	18000 J	20000 J	-	-

Notes: Units in mg/kg.

J = Estimated Value.

U = Not detected/below detection limit.

^(a) LECs are based on U.S. EPA Region 10 Microtox results (Cubbage and Breidenbach, 1994)

^(b) OMEE LELs are sediment quality guidelines developed by the Ontario Ministry of Environment and Energy (Persaud et al., 1993).

^(c) AVS and SEM values are presented in $\mu\text{mole/g}$.

TABLE 4.6-4
RESULTS OF THE IWW DITCH OUTFALL 10-DAY BIOASSAYS CONDUCTED OCTOBER, 1994

Test Species/Location	10-Day Survival (%)	Mean (\pm S.D.) Dry Weight (mg/surviving organism)
<i>C. tentans</i> ^(a)		
Control	79	1.54 \pm 0.46
SD16 (downstream of outfall)	96	1.97 \pm 0.15
SD17 (at outfall)	92	1.48 \pm 0.19
SD21 (upstream of outfall)	83	1.86 \pm 0.23
<i>H. azteca</i> ^(b)		
Control	95	0.34 \pm 0.03
SD16 (downstream of outfall)	90	0.32 \pm 0.03
SD17 (at outfall)	88	0.33 \pm 0.03
SD21 (upstream of outfall)	98	0.33 \pm 0.05

Notes: ^(a) Tests were conducted according to EA protocol ATS-SAI-HA-00, based on ASTM E1383-90. No statistical differences between groups ($p \leq 0.05$), using Tukey's Multiple Comparison method, was found for either survival or growth.

^(b) Tests were conducted according to EA protocol ATS-SAI-CT-00, based on ASTM E1383-90. No statistical differences between groups ($p \leq 0.05$), using Tukey's Multiple Comparison method, was found for either survival or growth.

TABLE 4.6-5
MEAN DELTA SEDIMENT CONCENTRATIONS OF COPCS IN MG COPC/KG BULK SEDIMENT WEIGHT

COPC	Mean Concentration – Portneuf River Delta^(a)	Mean Concentration – Snake River Delta^(a)	Mean Concentration – Upstream of EMF Facilities^(b)	Maximum Concentration – Portneuf River Delta	LEC^(c)	OMEE – LEL^(d)
Arsenic	2.8	3.1	4.3	4.6	40	6
Cadmium	0.93	0.37	NC	1.6	7	0.6
Flouride	350	250	330	530	-	-
Mercury	NC	NC	NC	0.46	0.56	0.2
Selenium	0.81	0.62	NC	1.7	-	-
Zinc	43	35	42	69	490	120

- Notes: ^(a) Mean concentrations for the Portneuf and Snake deltas are calculated from 10 sampling stations per river. At each sampling station one mudflat sample and one channel sample was taken.
- ^(b) Mean concentrations for samples collected in the Portneuf River upstream of the EMF facilities are based on 7 samples for all COPCs, except flouride, for which a high outlier was removed.
- ^(c) LECs are based on U.S. EPA Region 10 Microtox results (Cubbage and Breidenbach, 1994).
- ^(d) OMEE LELs are sediment quality guidelines developed by the Ontario Ministry of Environment and Energy (Persaud et al., 1993).
- ^(e) A mean sediment concentration was not calculated as over half of the samples were non-detects.

TABLE 4.6-6
COPC AND METALS CONCENTRATIONS IN PORTNEUF RIVER DELTA SEDIMENTS

Parameter (mg/kg)	Portneuf River Delta Channel Stations										Portneuf River Delta Bank Stations										Representative Soil Level
	PDC1	PDC2	PDC3	PDC4	PDC5	PDC6	PDC7	PDC8	PDC9	PDC10	PDB1	PDB2	PDB3	PDB4	PDB5	PDB6	PDB7	PDB8	PDB9	PDB10	
Aluminum, total	8160	8910	8390	5400	4990	5260	5130	6470	6080	6180	4610	10700	12500	12900	15100	14200	8210	5930	5300	7510	13,900
Arsenic, total	3.3	3.4	2.8	2.1	2.0	1.9	2.6	2.1	2.6	2.3	2.4	3.5	3.9	3.3	4.4	4.6	2.7	2.5	2.5	2.9 U	7.7
Cadmium, total	0.83	1.2	0.87	0.61	0.68	0.75	1.1	1.1	0.9	0.77	0.51	1.4	1.3	1.6	1.2	0.96	0.71	0.6	0.81	0.77	1.9
Fluoride, total	429	352	318	413	406	367	277	298	402	306	324	350	309	529	304	269	250	343	328	330	600
Iron, total	10000	10600	10200	6860	5940	7020	6320	8850	8780	8370	7080	11600	14300	14700	16700	18000	9240	8020	7430	10100	14400
Mercury, total	0.2 U	0.14 U	0.14 U	0.1 U	0.1 U	0.08 U	0.06 U	0.08 U	0.07 U	0.08 U	0.07 U	0.18 U	0.19 J	0.46	0.16 U	0.09 U	0.07 U	0.06 U	0.06 U	0.09 U	0.16
Selenium, total	1	0.72	1.1	0.65	0.42 U	0.37	0.77	0.54	0.52	0.69	0.61	0.73	1.7	1.5	1.6	1.3	0.95	0.38 U	0.59	0.5	3.05
Zinc, total	44.7	49.1	45.3	28.0	28.9	29.4	28.8	36.0	37.0	34.2	27.7	58.1	66.9	67.4	68.8	65.8	41	29.6	29.2	41.1	52.8

Notes: J = Estimated Value.
U = Not detected/below detection limit.

TABLE 4.6-7
CONCENTRATIONS OF NUTRIENTS AND OTHER PARAMETERS IN PORTNEUF RIVER DELTA SEDIMENTS

Parameter (mg/kg)	Portneuf River Delta Channel Stations										Portneuf River Delta Bank Stations									
	PDC1	PDC2	PDC3	PDC4	PDC5	PDC6	PDC7	PDC8	PDC9	PDC10	PDB1	PDB2	PDB3	PDB4	PDB5	PDB6	PDB7	PDB8	PDB9	PDB10
AVS	114 J	44.6 J	289 J	220 J	17.1 J	23.5 J	<8.1 J	<7.9 J	54.9 J	<8.7 J	<6.5 J	<6.8 J	<7.4 J	<7.5 J	<7.3 J	<6.8 J	<6.9 J	6.4 J	<6.8 J	<6.3 J
% Clay	5	3	1	2	0	0	0	0	0	0	0	4	5	5	10	10	6	0	0	1
SEM - Cd	0.26 UJ	0.38 J	0.31 J	0.20 J	0.13 J	0.13 J	0.13 J	0.13 J	0.23 J	0.20 J	0.07 UJ	0.36 J	0.17 J	0.52 J	0.21 J	0.22 J	0.17 J	0.10 J	0.12 J	0.20 J
SEM - Hg	1.2E-3 J	8.4E-4 J	1.0E-3 J	8.2E-4 J	1.8E-3 UJ	7.7E-4 J	8.1E-4 J	7.9E-4 J	7.9E-4 J	8.7E-4 J	6.5E-4 J	6.0E-3 J	7.4E-4 J	1.2E-2 J	2.1E-3 UJ	3.0E-3 UJ	2.6E-3 UJ	2.4E-3 UJ	3.2E-3 UJ	4.6E-3 UJ
SEM - Zn	9.54 J	9.86 J	10.3 J	7.03 J	5.47 J	5.31 J	5.21 J	5.19 J	8.51 J	8.12 J	3.32 J	9.34 J	5.57 J	13.3 J	6.81 J	10.9 J	7.10 J	4.72 J	4.88 J	6.46 J
TOC	61800	45000	45900	40200	56400	67600	28200	28100	38000	55700	37100	24500	37100	24500	42300	48800	48100	61600	52300	40100
pH	N/A	8.13	N/A	N/A	8.12	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Conductiv. ^(a) (mS/cm)	N/A	0.557	N/A	N/A	0.559	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Turbidity ^(a) (NTU)	N/A	4	N/A	N/A	5	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Diss. O ₂ ^(a) (mg/L)	N/A	5.5	N/A	N/A	7.88	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Temp.(°C) ^(a)	N/A	9.7	N/A	N/A	6.2	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Salinity ^(a) (‰)	N/A	0.02	N/A	N/A	0.02	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A

Notes: J = Estimated value.
N/A = Not available.
U = Not detected/below detection limit.
(a) = Measurements on surface water at the sediment sampling location.

TABLE 4.6-8
COPC AND METALS CONCENTRATIONS IN SNAKE RIVER DELTA SEDIMENTS

Parameter (mg/kg)	Snake River Delta Channel Stations										Snake River Delta Bank Stations										Representative Soil Levels
	SDC1	SDC2	SDC3	SDC4	SDC5	SDC6	SDC7	SDC8	SDC9	SDC10	SDB1	SDB2	SDB3	SDB4	SDB5	SDB6	SDB7	SDB8	SDB9	SDB10	
Aluminum, total	8740	11500	2020	4420	5130	4070	3430	3180	2720	1950	4170	4030	4010	5340	7990	7790	6490	2420	5980	5560	13900
Arsenic, total	4.5	11.3	1.9	2.5	2.8	2.1	2.1	2.2	2.0	1.8	1.7	1.9	2.1	2.4	3.5	3.2	2.9	2.8	5.4	3.0	7.7
Cadmium, total	0.70	0.79	0.27 U	0.32 U	0.40	0.32	0.23 U	0.22	0.28 U	0.24 U	0.36	0.30	0.25	0.39	0.63	0.61	0.62	0.19	0.51	0.42	1.9
Fluoride, total	301	238	114	270	253	259	242	183	188	140	381	184	187	282	268	327	329	185	380	210	600
Iron, total	12100	19000	4900	7100	9470	7440	6630	6460	5550	4630	7030	6720	6650	8290	10900	12100	10900	5660	12100	9280	14400
Mercury, total	0.07 U	0.07 U	0.05 U	0.07 U	0.08 U	0.06 U	0.06 U	0.05 U	0.06 U	0.06 U	0.05 U	0.05 U	0.07 U	0.06 U	0.08 U	0.07 U	0.06 U	0.05 U	0.16 U	0.05 U	0.16
Selenium, total	0.82	0.85	0.40 U	0.74	0.7	0.57	0.46	0.43	0.64	0.43	0.58	0.48	0.56	0.51	1.1	0.98	0.65	0.30	0.83	0.61	3.05
Zinc, total	51.1	70.9	19.9	32	37	27.3	26.1	25.5	25.7	18.9	34.2	27.2	26.9	34.1	50.6	50.5	42.4	20.5	46.0	37.2	52.8

Notes: U = Not detected/below detection limit.

TABLE 4.6-9
CONCENTRATIONS OF NUTRIENTS AND OTHER PARAMETERS IN SNAKE RIVER DELTA SEDIMENTS

Parameter (mg/kg)	Snake River Delta Channel Stations										Snake River Delta Bank Stations									
	SDC1	SDC2	SDC3	SDC4	SDC5	SDC6	SDC7	SDC8	SDC9	SDC10	SDB1	SDB2	SDB3	SDB4	SDB5	SDB6	SDB7	SDB8	SDB9	SDB10
AVS	<8.3 J	<7.5 J	<6.7 J	<7.7 J	258 J	156 J	18 J	<7.0 J	<7.1 J	<7.1 J	<6.7 J	<6.4 J	<6.3 J	<6.7 J	<8.4 J	<7.1 J	<6.7 J	<6.2 J	9.6 J	<5.5 J
% Clay	10	27	0	0	3	0	0	0	0	0	0	2	3	7	4	5	4	0	5	3
SEM - Cd	0.23 J	0.24 J	0.17 UJ	0.16 UJ	0.18 UJ	0.12 UJ	0.05 UJ	0.06 UJ	0.02 UJ	0.02 UJ	0.12 J	0.10 UJ	0.04 UJ	0.16 UJ	0.21 UJ	0.28 J	0.21 J	0.02 UJ	0.21 UJ	0.19 UJ
SEM - Hg	8.3E-4 J	2.9E-3 J	6.7E-4 J	7.7E-4 J	8.3E-4 J	7.2E-4 J	7.1E-4 J	7.0E-4 J	7.1E-4 J	7.1E-4 J	6.7E-4 J	6.4E-4 J	1.3E-3 J	6.7E-4 J	8.4E-4 J	2.1E-3 J	6.7E-4 J	6.2E-4 J	9.6E-4 J	1.7E-3 J
SEM - Zn	10.9 J	11.8 J	2.57 J	7.01 J	9.48 J	6.56 J	5.75 J	4.79 J	5.30 J	4.01 J	6.24 J	5.16 J	4.66 J	7.38 J	12.39 J	11.91 J	9.84 J	4.71 J	14.65 J	7.72 J
TOC	44000	22000	9880	30600	37500	20100	46300	25900	12000	20600	24300	14300	14000	16600	47300	41700	45600	12400	53400	28000
pH	N/A	N/A	N/A	7.78	8.14	8.03	7.93	N/A	8.12	8.09	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Conductivity (mS/cm)	N/A	N/A	N/A	0.401	0.410	0.397	0.394	N/A	0.401	0.389	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Turbidity (NTU)	N/A	N/A	N/A	6	10	8	2	N/A	2	1	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Diss. O ₂ (mg/l.)	N/A	N/A	N/A	8.39	8.48	10.54	10.77	N/A	9.79	10.51	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Temp. (°C)	N/A	N/A	N/A	9.1	8.8	9.8	9.6	N/A	7.6	8.5	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Salinity (%)	N/A	N/A	N/A	0.01	0.01	0.01	0.01	N/A	0.01	0.01	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A

Notes: Units in mg/kg.
J = Estimated value.
U = Not detected/below detection limit.

TABLE 4.6-10

MEAN (\pm STANDARD DEVIATION) AND MEDIAN CONCENTRATIONS OF SOIL PARAMETERS IN THE SAGEBRUSH STEPPE HABITAT^(a)

Site		Cd	F	Zn	Na	K	Mg	Fe	Ca	TOC	pH ^(b)	CEC ^(c)
Bannock Hills	Mean \pm S.D.	27 \pm 5.0	1400 \pm 230	260 \pm 47	6.0 \pm 2.6	7.5 \pm 4.4	3.9 \pm 2.5	3.9 \pm 3.9	37 \pm 23	10000 \pm 1700	7.68 - 7.94	25 \pm 2.3
	Median	28	1400	260	5.7	5.0	2.3	2.2	24	10000	7.83	25
Michaud Flats	Mean \pm S.D.	21 \pm 7.7	1800 \pm 720	160 \pm 49	6.4 \pm 1.5	10 \pm 4.5	8.5 \pm 4.6	16 \pm 8.4	26 \pm 6.7	16000 \pm 4400	6.51 - 7.38	27 \pm 3.6
	Median	21	1700	170	5.9	8.5	6.6	11	25	15000	7.06	27
Ferry Butte	Mean \pm S.D.	0.49 \pm 0.19	360 \pm 31	56 \pm 4.6	6.5 \pm 1.4	20 \pm 8.7	17 \pm 9.8	36 \pm 29	24 \pm 9.4	10000 \pm 8200	6.62 - 7.75	23 \pm 2.0
	Median	0.46	360	57	6.7	21	16	24	21	8200	6.98	22

Notes: (a) All reported values are in mg/kg unless otherwise noted. All means and standard deviations are based upon 10 samples.

(b) pH values are presented as a range.

(c) Cation-exchange capacity is reported in mEq/100g.

TABLE 4.6-11

MEAN (\pm STANDARD DEVIATION) AND MEDIAN CONCENTRATIONS OF SOIL PARAMETERS IN THE RIPARIAN HABITAT^(a)

Site		Cd	F	Zn	Na	K	Mg	Fe	Ca	TOC	pH ^(c)	CEC ^(d)
Portneuf River	Mean \pm S.D.	10 \pm 8.5	1100 \pm 760	110 \pm 49	85 \pm 50	32 \pm 26	31 \pm 31	4.1 \pm 4.9	71 \pm 72	22000 \pm 6300	8.6 - 7.2	43 \pm 14
	Median	6.6	835	82	71	21	18	2.2	43	22000	8.2	41
Snake River	Mean \pm S.D.	0.19 \pm 0.02 ^(b)	240 \pm 34	23 \pm 75	8.9 \pm 2.2	15 \pm 5.8	5.8 \pm 1.7	3.1 \pm 2.1	27 \pm 6.0	13000 \pm 6000	8.0 - 7.7	17 \pm 4.6
	Median	0.20	250	24	8.1	14	5.7	3.1	27	11000	7.9	16

Notes: ^(a) All reported values are in mg/kg unless otherwise noted. All means and standard deviations are based upon 10 samples.

^(b) 8/10 samples were detects. One-half of the Sample Quantitation Limit (SQL), as reported, was used in place of a measured concentration to calculate summary statistics.

^(c) pH values are presented as a range rather than a mean and standard deviation.

^(d) Cation-exchange capacity is reported in mEq/100g.

TABLE 4.6-12A
METAL COPC CONCENTRATIONS DETECTED IN THE TERRESTRIAL SOIL SAMPLE (BANNOCK HILLS)
SELECTED FOR MINERALOGICAL FEASIBILITY ANALYSIS

Particle Size Fraction	Arsenic (mg/kg)	Cadmium (mg/kg)	Zinc (mg/kg)
≤ 2.0 mm	36	24	376
≤ 500 μm	36	35	456
≤ 250 μm	35	43	445
≤ 125 μm	18	25	292
≤ 63 μm	19	20	275

TABLE 4.6-12B
METAL COPC CONCENTRATIONS DETECTED IN THE SLAG SAMPLE SELECTED FOR MINERALOGICAL
FEASIBILITY ANALYSIS

Particle Size Fraction	Arsenic (mg/kg)	Cadmium (mg/kg)	Zinc (mg/kg)
≤ 2.0 mm	61	14	736
≤ 500 μm	28	31	987
≤ 250 μm	50	77	1500
≤ 125 μm	-*	-	-
≤ 63 μm	-	-	-

Notes *Fraction does not exist.

TABLE 4.6-13

MEAN (\pm STANDARD DEVIATION) AND MEDIAN CONCENTRATIONS OF CADMIUM, FLUORIDE AND ZINC IN VEGETATION OF THE SAGEBRUSH STEPPE HABITAT^(a)

Site		Cd - Sagebrush ^(b) (unwashed)	Cd - Sagebrush ^(b) (washed)	Cd - Grass ^(c)	F - Sagebrush ^(b) (unwashed)	F - Grass ^(c)	Zn - Sagebrush ^(b) (unwashed)	Zn - Sagebrush ^(b) (washed)	Zn - Grass ^(c)
Bannock Hills	Mean \pm S.D.	0.99 \pm 0.12	0.77 \pm 0.17	0.54 \pm 0.18	74 \pm 30 ^(d)	62 \pm 28	31 \pm 4.1	26 \pm 3.3	11 \pm 3.3
	Median	1.0	0.76	0.50	69	49	31	26	12
Michaud Flats	Mean \pm S.D.	1.3 \pm 0.26	1.1 \pm 0.24	0.46 \pm 0.08	52 \pm 24 ^(e)	22 \pm 15 ^(f)	38 \pm 5.4	33 \pm 8.9	11 \pm 2.6
	Median	1.2	1.1	0.47	54	12	37	35	11
Ferry Butte	Mean \pm S.D.	0.17 \pm 0.09 ^(g)	0.17 \pm 0.09 ^(f)	0.12 \pm 0.10 ^(h)	12 \pm 0.29 ⁽ⁱ⁾	12 \pm 0.23 ⁽ⁱ⁾	30 \pm 6.0	28 \pm 4.9	8.2 \pm 1.4
	Median	0.15	0.10	0.10	12	12	29	26	8.2

Notes: (a) All reported values are in mg/kg. All means and standard deviations are based upon 10 samples, except for fluoride concentrations in big sagebrush which are based on 20 samples per site. All grass samples are unwashed. [When samples were below the detection limit, one-half of the Sample Quantitation Limit (SQL), as reported, was used in place of each measured concentration to calculate summary statistics.]

(b) Sagebrush refers to big sagebrush.

(c) Grass refers to thickspike wheatgrass.

(d) 18/20 samples were detects.

(e) 19/20 samples were detects.

(f) 4/10 samples were detects.

(g) 5/10 samples were detects.

(h) 2/10 samples were detects.

(i) No samples were detects.

Missing
Tux

4.6.12B

TABLE 4.6-14
MEAN (\pm STANDARD DEVIATION) AND MEDIAN CONCENTRATIONS OF CADMIUM, FLUORIDE AND ZINC
IN RUSSIAN OLIVES OF THE RIPARIAN HABITAT^(a)

Site		Cd - Olive	F - Olive	Zn - Olive
Portneuf River	Mean \pm S.D.	0.18 \pm 0.09 ^(b)	12 \pm 0.47 ^(c)	10 \pm 1.8
	Median	0.15	12	10
Snake River	Mean \pm S.D.	0.16 \pm 0.18 ^(d)	12 \pm 0.37 ^(c)	7.2 \pm 1.3
	Median	0.10	12	7.3

Notes: ^(a) All reported values are in mg/kg. All samples are unwashed. [When samples were below the detection limit, one-half of the Sample Quantitation Limit (SQL), as reported, was used in place of each measured concentration to calculate summary statistics.]

^(b) 4/10 samples were detects.

^(c) 0/10 samples were detects.

^(d) 1/10 samples were detects.

TABLE 4.6-15
SOIL-TO-PLANT UPTAKE FACTORS (B VALUES)
BASED ON VEGETATION IN THE SAGEBRUSH STEPPE HABITAT^(a)

Plant Species	COPC	Bannock Hills	Michaud Flats	Ferry Butte	Site Average ^(b)	Baes Default ^(c)	Site/Baes ^(d)
Thickspike Wheatgrass (Unwashed)	Cd	0.0200	0.0219	0.2449	0.0210	—	—
	F	0.0443	0.0122	0.0333	0.0283	—	—
	Zn	0.0423	0.0688	0.1464	0.0555	—	—
Big Sagebrush (washed)	Cd	0.0285	0.0524	0.3469	0.0409	—	—
	Zn	0.1000	0.2063	0.5000	0.1531	—	—
Big Sagebrush (unwashed)	Cd	0.0367	0.0619	0.3469	0.0493	—	—
	F	0.0529	0.0289	0.0333	0.0409	—	—
	Zn	0.1192	0.2375	0.5357	0.1784	—	—
Overall Average ^(e)	Cd	—	—	0.2959	0.0351	0.5500	0.0639
	F	—	—	0.0333	0.0346	0.0600	0.5761
	Zn	—	—	0.3411	0.1169	1.5000	0.0780

Notes: ^(a) B value is the ratio of COPC concentration in plant tissue to soil COPC concentration.

^(b) Site average is the average B values of the two potentially impacted plots (Bannock Hills and Michaud Flats).

^(c) Site/Baes is the ratio of the site-specific averages/Baes model default.

^(d) Default values developed by Baes et al. (1984).

^(e) Overall average is the average B values of unwashed big sagebrush and thickspike wheatgrass for the two potentially impacted plots.

TABLE 4.6-16
SOIL-TO-PLANT UPTAKE FACTORS (B VALUES) BASED ON RUSSIAN OLIVES OF THE RIPARIAN HABITAT^(a)

Plant Species	COPC	Portneuf River	Snake River	Baes Default ^(b)	Portneuf/Baes ^(c)
Russian Olives	Cd	0.0180	0.8158	0.1500	0.1200
	F	0.0109	0.0496	0.0060	1.8182
	Zn	0.0909	0.3106	0.9000	0.1010

Notes: ^(a) B value is the ratio of COPC concentration in plant tissue to soil COPC concentration.

^(b) Default values developed by Baes et al. (1984).

^(c) Portneuf/Baes is the ratio of the Portneuf River B values to the Baes Default values.

TABLE 4.6-17
MEAN (\pm STANDARD DEVIATION) AND MEDIAN MOUSE TISSUE CONCENTRATIONS IN THE
SAGEBRUSH STEPPE HABITAT^(a)

Location		Cd – Whole Body ^(b)	F – Whole Body ^(b)	Zn – Whole Body ^(b)	F – Femur ^(c)
Bannock Hills	Mean \pm S.D.	0.61 \pm 0.27	128.5 \pm 26.2	38.5 \pm 4.5	290.8 \pm 197.6
	Median	0.62	124	38.8	253
Michaud Flats	Mean \pm S.D.	0.22 \pm 0.13	90.9 \pm 30.3	37.6 \pm 3.8	633.3 \pm 220.3
	Median	0.16	87.5	36.5	585
Ferry Butte	Mean \pm S.D.	0.07 \pm 0.05	6.81 \pm 0.45 ^(d)	38.6 \pm 6.5	130 \pm 90 ^(e)
	Median	0.04	6.80	39.2	80

Notes: ^(a) Ten samples were collected for whole body and femur analysis from each of the three sampling locations. All values are in mg COPC/kg body weight. [When samples were below the detection limit, one-half of the Sample Quantitation Limit (SQL), as reported, was used in place of each measured concentration to calculate summary statistics.]

^(b) Whole body concentration (wet weight).

^(c) Concentration from both femurs (fresh weight).

^(d) 0/10 samples were detects.

^(e) 3/10 samples were detects.

4.7 AIR MONITORING

4.7.1 INTRODUCTION

Objectives

Both FMC and Simplot are active production facilities with regulated air emissions. The Air Pathways Program, which consisted of the ambient Air Pathways Monitoring Program and the Air Pathways Modeling Program, was established to evaluate off-site transport of site-derived constituents for the RI/FS investigation. The Air Pathways Monitoring Program provided site-specific ambient air concentration data which necessarily included the measurement of particulates associated with emissions both from EMF activities and from non-EMF activities. The Air Pathways Modeling Program was established to distinguish EMF sources from non-EMF sources. In the local environment, non-EMF sources could include natural sources such as local wind borne dust or emissions from brush fires, and anthropomorphic or man-made sources such as emissions associated with activities at the City of Pocatello's sewage treatment plant, wood stoves, or car exhaust. The Air Pathways Modeling Program report which includes the quantification of emissions from EMF sources and the likely site-related constituents associated with these sources is contained in Part III, Volume 2. This section of the RI report is limited to a discussion of the air monitoring results. A complete presentation of the air monitoring data is contained in Part III, Volume 1.

The monitoring program for the RI/FS investigation is described in the Air Pathways Monitoring Plan (Bechtel, 1993b and 1993e). The air monitoring program for the RI/FS investigation extended from October 1993 through October 1994. Samples were collected of total suspended particulate (TSP) and the PM₁₀ fraction. The PM₁₀ fraction is intended to measure particulate matter less than or equal to 10 microns in diameter, but may in fact capture particulates up to 25 microns in diameter. Nonetheless, PM₁₀ is a better indicator of the particles that have potential access to the lung than TSP, and is therefore more relevant to human health. PM₁₀ constituents were measured routinely only through March 1994, per agreement with the EPA. Initially, 22

chemical and radiological constituents were targeted for analysis in particulate samples. These included: aluminum, arsenic, barium, beryllium, cadmium, total chromium, nickel, particulate fluoride, phosphorus, selenium, silver, manganese, thallium, vanadium, zinc, lead-210, polonium-210, total uranium, thorium-230, thorium-232, radium-226, and radium-228. Gaseous fluoride and crystalline silica were also analyzed.

Ambient air screening levels were proposed for air constituents by EPA Region 10 (EPA, 1993b) and were used to evaluate sampling results. These screening values were estimated to represent a concentration which is health-protective. In February 1994, analyses for nine constituents were discontinued because the constituents were never detected or detected only at concentrations lower than the EPA's screening guidelines. Samples were analyzed for the remaining 12 PM₁₀ constituents until the end of the RI/FS air monitoring program (arsenic, cadmium, chromium, nickel, phosphorus, vanadium, zinc, lead-210, polonium-210, total uranium, thorium-230, and radium-226). Silica and total fluoride were also analyzed until the end of the RI/FS air monitoring program, but not in the PM₁₀ samples.

The air monitoring program established seven sampling stations (designated as Sites). One station was selected as representing an upwind background area (Site 6). Three stations were chosen to validate the air model representing the likely maximally impacted areas (Sites 1, 2, and 7). An additional three stations were chosen to represent potential impact to residential areas (Sites 3, 4, and 5).

Overview of Findings

The air monitoring stations at Sites 3, 4, and 5 represent the potential exposure points for the residents in the vicinity of the EMF facilities. The PM₁₀ mass arithmetic mean concentrations for Sites 3, 4, and 5 were compared to the concentrations for Site 6, the background site. The daily measurements of the mass of PM₁₀ at each of the three sites was determined to closely correlate to the background site values. The nature and extent of potential residential exposure was assessed by comparing the average PM₁₀ constituent concentrations to the background

concentration and to the EPA Region 10 screening levels. The average concentration is used because the screening levels provided by EPA are associated with chronic, long-term exposure and therefore are not appropriate for comparing daily monitoring results. Background concentrations are based on the 95th percentile of Site 6 concentrations for each constituent. The 95th percentile is used to represent the upper end of the range of background concentrations. The daily sampling results are discussed in their entirety in Part III, Volume 1.

Most PM₁₀ average constituent concentrations at Sites 3, 4, and 5 were present at concentrations below screening levels or within the range of background concentrations. Only cadmium, chromium, and phosphorus concentrations were above both the EPA screening levels and the 95th percentile of background concentrations such that:

- Cadmium concentrations were above both the screening level and background concentration at Site 5 but were never more than two-fold greater;
- Phosphorus concentrations were above both the screening level and background concentration at Sites 3 and 5 but were never more than two-fold larger than the screening value and three-fold greater than background;
- Chromium, measured as total chromium, was above both the screening level and background concentration at Sites 3, 4, and 5. However, the screening level is based on the chromium (VI) species as opposed to total chromium. When the total chromium concentrations are adjusted for the maximum estimated chromium (VI) concentration (Bechtel, 1994j), the average concentrations did not exceed the Region 10 screening value.

Section Organization

The organization of the remainder of this section is as follows: Section 4.7.2 describes the air monitoring program. Section 4.7.3 discusses the results of the air monitoring program at stations with potential human population exposure. Section 4.7.4 summarizes the results of the analysis of the air monitoring data. Additional detailed data from the RI/FS air monitoring program are discussed in Part III, Volume 1. The related air modeling studies are presented in Part III, Volume 2.

4.7.2 MONITORING PROGRAM

Monitoring Stations

The rationale for locations of air monitoring stations was described in the Air Pathways Monitoring Plan (Bechtel, 1993b and 1993e). Seven sampling stations were established for the air monitoring program. Stations were selected based on preliminary modeling results and meteorological data. Sites 1, 2, and 7 were chosen to validate air modeling. These sites on the boundary of the EMF plant areas were designated “perimeter” Sites and selected as representative of the likely maximally impacted areas based on preliminary modeling results. Sites 1 and 7 had previously been established as Simplot meteorological monitoring sites. Site 2 is in the highway right-of-way and neighboring a dirt road therefore unsuitable for estimating residential exposure. The location of Site 2 does not comply with EPA’s Ambient Monitoring Guidelines, and therefore data collected at Site 2 do not represent “ambient air”. Furthermore, Site 2 was roughly one-half mile downwind of BAPCO, a significant source of fugitives, which is now shut down. Sampling occurred when this paving company was active.

The locations of the air monitoring stations are presented in Figure 4.7-1. Site 6 was selected as a background location. Site 6 is located 12 miles from the EMF site in the predominantly upwind direction. Sites 3, 4, and 5 were selected to study potential air impacts in residential areas. Sites 3, 4, and 5 are located from 1 to 3 miles from the site in areas with human receptors and designated as “community” sites. Site 3 is located in a residential area approximately two miles northeast and downwind of the plants. Site 4 is located one and a half miles to the east of the plants, near a mobile home park. Site 5 is located two miles southwest and in a direction predominately upwind of the plants on Tribal land.

Analysis

Sampling consisted of the collection of PM₁₀ and TSP particulate filters every second day. PM₁₀ is regulated under the National Ambient Air Quality Standards (NAAQS) (EPA, 1987a). TSP is regulated by the State of Idaho (State of Idaho, 1994). Every fourth day, PM₁₀ and TSP filters

were subject to routine chemical and radiological analyses. As recommended by EPA, samples that were not subject to routine analysis were first analyzed for PM₁₀ and TSP concentrations; if the TSP concentration exceeded 200 µg/m³ (80% of the Idaho 24-hour standard) or the PM₁₀ concentration exceeded 120 µg/m³ (80% of the NAAQS 24-hour standard), the filters were then subject to chemical and radiological analyses (Bechtel, 1993b, 1993e, and 1994b).

Twelve PM₁₀ constituents were carried through the entire air monitoring program. The analytes included: arsenic, cadmium, chromium, nickel, phosphorus, vanadium, zinc, lead-210, polonium-210, total uranium, thorium-230, and radium-226. The analyses were discontinued for constituents that were not detected or detected only at daily concentrations (i.e., measured over a 24-hour period) lower than the screening levels.

Screening Levels

EPA Region 10 provided screening levels to be used for comparison to the monitoring data. The ambient air screening levels are designed to be risk-based concentrations which are protective of residents who may be exposed chronically, i.e., continuously over a long period. The screening levels are based on many protective assumptions with ample margins of safety. Screening levels are calculated so concentrations below screening levels are clearly not of public health concern and no further analysis is required. The screening levels are not health-based standards and do not indicate that any concentration exceeding screening levels produces health effects.

Consequently, for a comparison of the potential human health effect, the screening levels should be compared to the long-term average exposures as represented by the average concentrations measured in the monitoring program. Furthermore, the PM₁₀ fraction is a conservative approximation of the fraction of particulate matter that may be inhaled and therefore only analyte concentrations from the PM₁₀ fraction are appropriate for comparison to the screening levels. More detail on the development of these screening levels is provided below for those analytes monitored throughout the air monitoring program.

Arsenic, cadmium, chromium, and nickel. The reference for the EMF screening levels for arsenic, cadmium, chromium, and nickel is the EPA Region III Risk-Based Concentration (RBC) Table, January 1993 (EPA, 1993a). The concentrations are based on inhalation slope factors, however these concentrations assumed total chromium measured was 100% chromium (VI).

When released to the environment, the vast majority of chromium (VI) is rapidly converted to the more stable and less toxic chromium (III). It is likely that chromium (VI) would not be detectable in the PM₁₀ samples because the concentrations are so low. For processes at the EMF site, an analysis of chromium species estimated chromium (VI) to be less than 1% of the total chromium based on chromium (VI) measurements in two crushed ferrophos samples (Bechtel, 1994j). Therefore, to estimate chromium (VI) for screening purposes, chromium (VI) was assumed to be 1% of the total chromium concentration. Similarly, nickel is evaluated as smelting fumes, although the EMF processes do not include smelting. This adds to the uncertainty of the nickel estimate for screening purposes.

Vanadium, fluoride, and phosphorus. EPA Region 10 ambient air screening levels for vanadium, fluoride, and phosphorus are based on the State of Washington Administrative Code (WAC) class B toxic air pollutants (WAC, 1991). (Vanadium is only listed as vanadium pentoxide, V₂O₅, in the WAC.) According to the WAC (173-460-110, number 2 a and b), in the absence of an EPA reference concentration, the threshold limit value - time weighted average (TLV-TWA) was divided by 300 to calculate a TWA which could be considered a chronic exposure screening concentration.

Silica. An EPA Region 10 screening value for silica was not determined at the initiation of the air monitoring program.

Zinc. An EPA Region 10 screening value for zinc was not determined at the initiation of the air monitoring program. For comparison purposes, EPA Region 3 RBC of $1.1 \times 10^3 \mu\text{g}/\text{m}^3$

was used although this value is not based on inhalation of zinc, but rather ingestion. Therefore, this RBC for zinc may overestimate inhalation risk, but studies are not available to evaluate inhalation risk directly.

Radionuclides. EPA Region 10 ambient air screening levels for radionuclides reference the Health Effects Assessment Summary Tables (HEAST). All of the values in HEAST are based on chronic inhalation (EPA, 1994). EPA Region 10 provided screening levels for three of the uranium isotopes (uranium-234, uranium-235, and uranium-238). For screening purposes, the most conservative uranium (uranium-238) value provided by EPA Region 10 was used.

4.7.3 MONITORING RESULTS

Data Treatment

Only Sites 3, 4, and 5 represent potential residential exposure stations such that only these Sites are appropriate for comparison to the ambient air screening levels. Sites 1, 2, and 7 are facility perimeter sites which were established to validate the dispersion model. They are located on nonresidential properties. Site 1 is on company-owned land, Site 2 is located on a highway right-of-way transecting company-owned land, and Site 7 is on BLM land. Comparison of the data from these perimeter sites to residential screening levels is not relevant to current or future potential exposures. In the following analysis, the residential ambient air screening levels are compared to measured concentrations only at Sites 3, 4, and 5. A detailed evaluation of the data from the perimeter sites, as well as all sample results, is provided in Part III, Volume 1.

Data for PM₁₀ mass analysis were collected from October 1993 to October 1994. Data for PM₁₀ constituent concentrations used in the analysis of air monitoring results were limited to the period between October 1993 to March 1994. Sample data for all stations from two dates, 10/04/93 and 11/25/93, were excluded. On the first date, potato harvest trucks stirred up large amounts of particles and on the second date anomalous wind conditions caused Site 6, the background location, to be downwind from the plant areas. Rejected data with the R qualifier

were eliminated, and blank contaminated samples (designated in the database with a 6 or 7 in “qual_code” field) were treated as recommended in guidance for risk assessment (EPA, 1989). Data designated as Site 9 are co-located samples taken at Site 3, and these data were averaged for samples taken on the same date. Likewise, data designated as Site 8 are co-located samples taken at Site 1, and these data were averaged.

For the PM₁₀ mass analysis, monitoring data accompanied by the R field qualifiers, indicating that the sample results were rejected due to quality assurance validation criteria, were not included in the analysis. A more detailed description of these qualifiers as they relate to specific sampling results is provided in Part III, Volume 1.

The data were analyzed statistically using a proprietary statistical analysis program (SIRIS). In this program, nondetects (U flags) were replaced by one-half the detection limit (defined in Part III, Volume 1), high nondetects (samples with one-half the detection limit greater than the maximum detected value) were eliminated, and duplicates were averaged, in accordance with EPA's guidance (EPA, 1989). Statistical analyses, including calculation of the 95th percentile, to characterize background concentrations, were performed according to the methods described in Gilbert (1987). If all samples had undetectable concentrations of a constituent the values presented are one-half the detection limit and are labeled “U” in the data tables. If there was only one detected concentration of a constituent, this was used in place of the 95th percentile and labeled with an “M” for measured.

Results: PM₁₀ Mass

The average concentrations of PM₁₀ mass at Sites 3, 4, and 5 were compared to the average background concentration of PM₁₀ at Site 6 (Table 4.7-1). The background level has an arithmetic mean value of 19.4 µg/m³. The primary NAAQS standard for PM₁₀ is an annual arithmetic mean of 50 µg/m³ calculated by averaging quarterly values over three years (EPA, 1987a).

Among the sites to be considered for potential impact to populated areas (Sites 3, 4, and 5), the measured mass of PM_{10} were similar. Among all measurements made at these three sites, the arithmetic mean of the concentrations was much less than the NAAQS standard of $50 \mu g/m^3$. The fact that levels at these sites were often below background is reflected in the fact that the highest concentrations measured at each of the three sites were all below the three highest levels measured at Site 6.

Comparing PM_{10} levels measured at each site to simultaneous measurements at the background station, Site 6, revealed that there was a high degree of correlation with background. The correlation between daily sampling results at each site (3, 4, and 5) to the measured background level was significant at greater than the 99.9% level, based on a calculation of the t-statistic as determined from the correlation coefficient and number of sampling days. This strong correlation indicates that the residential sites are being primarily influenced by the same factors that affect the background, Site 6.

Results: PM_{10} Constituents

Indicative of chronic exposure, the average concentration of each of the twelve analytes at each of Sites 3, 4, and 5 (Tables 4.7-3 through 4.7-5) was compared to three criteria. The average concentrations at each site were compared to the Region 10 ambient air screening levels. Secondly, the background average concentration was used for comparison. Additionally, the background 95th percentile concentration was used for comparison as a representation of the upper end of the background range of concentrations. The comparisons at each site location are provided below preceded by a discussion of the representative background concentrations.

Background, Site 6

The frequency of detection, range of concentrations detected, average, and 95th percentile of concentrations for each PM_{10} constituent at Site 6 are presented in Table 4.7-2. As shown in this table, several of the concentrations measured at Site 6, the representative background concentration, exceed EPA Region 10 screening values. The average background concentrations

of lead-210 and polonium-210, and the 95th percentile background concentrations of arsenic, chromium, lead-210, polonium-210, radium-226, and uranium, exceeded the Region 10 proposed screening values. In Table 4.7-2, the background average and 95th percentile concentrations are highlighted if they exceed the Region 10 screening levels.

Site 3

Table 4.7-3 summarizes the air monitoring results for Site 3 and compares the results to both Site 6, the background location, and the Region 10 screening values. In Table 4.7-3 background and screening values are highlighted if they are exceeded by the average analyte concentration for Site 3.

Arsenic. The average arsenic PM_{10} concentration at Site 3 was at the Region 10 screening level. Compared to background, the average concentration of arsenic at Site 3 was slightly higher than the background average concentration but less than one-half the background 95th percentile concentration.

Cadmium. The average cadmium PM_{10} concentration at Site 3 was below the Region 10 screening level. Compared to background, the average concentration of cadmium at Site 3 was higher than the background average concentration, and slightly greater than the background 95th percentile concentration.

Chromium. The average total chromium PM_{10} concentration at Site 3 [$9.24 \times 10^{-4} \mu\text{g}/\text{m}^3$] was above the Region 10 chromium (VI) screening level [$2.00 \times 10^{-4} \mu\text{g}/\text{m}^3$]. Converting the average concentration of total chromium at Site 3 to the maximum percentage estimated to exist as chromium (VI), the value is $9.24 \times 10^{-6} \mu\text{g}/\text{m}^3$. This value is less than the Region 10 chromium (VI) screen of $2.00 \times 10^{-4} \mu\text{g}/\text{m}^3$. Compared to background, the average concentration of chromium at Site 3 was greater than both the background average concentration [$1.34 \times 10^{-4} \mu\text{g}/\text{m}^3$] and the background 95th percentile concentration [$6.58 \times 10^{-4} \mu\text{g}/\text{m}^3$].

Nickel. The average nickel PM_{10} concentration at Site 3 was below the Region 10 screening level. Compared to background, the average concentration of nickel at Site 3 was higher than the background average concentration but below the single detected background concentration.

Phosphorus. The average phosphorus PM_{10} concentration at Site 3 [$3.22 \times 10^{-1} \mu\text{g}/\text{m}^3$] was 10 percent greater than the Region 10 screening level [$3.00 \times 10^{-1} \mu\text{g}/\text{m}^3$]. Compared to background, the average concentration of phosphorus was higher than both the background average concentration [$3.34 \times 10^{-2} \mu\text{g}/\text{m}^3$] and the background 95th percentile concentration [$2.03 \times 10^{-1} \mu\text{g}/\text{m}^3$].

Vanadium. The average vanadium PM_{10} concentration at Site 3 was below the Region 10 screening level. Compared to background, the average concentration of vanadium at Site 3 was higher than the background average concentration but lower than the background 95th percentile concentration.

Zinc. The average zinc PM_{10} concentration at Site 3 was below the EPA Region III screening level. (Region 10 did not provide a screening level for zinc.) Compared to background, the average concentration of zinc at Site 3 was higher than both the background average concentration and the background 95th percentile concentration.

Lead-210. The average lead-210 PM_{10} concentration at Site 3 [$2.51 \times 10^{-2} \text{pCi}/\text{m}^3$] was greater than the Region 10 screening level [$1.20 \times 10^{-3} \text{pCi}/\text{m}^3$]. However, the average background lead-210 concentration was also above the Region 10 screening level. Compared to background, the average concentration of lead-210 at Site 3 was 10 percent greater than the background average concentration [$2.35 \times 10^{-2} \text{pCi}/\text{m}^3$] but less than one-half the 95th percentile background concentration [$5.41 \times 10^{-2} \text{pCi}/\text{m}^3$].

Polonium-210. The average polonium-210 PM_{10} concentration at Site 3 [$1.25 \times 10^{-2} \text{pCi}/\text{m}^3$] was above the Region 10 screening level [$1.80 \times 10^{-3} \text{pCi}/\text{m}^3$]. However, the average

background polonium-210 concentration was also above the Region 10 screening level. Compared to background, the average concentration of polonium-210 at Site 3 was higher than the background average concentration [6.33×10^{-3} pCi/m³] but almost one-half the 95th percentile background concentration [2.05×10^{-2} pCi/m³].

Radium-226. Radium-226 was not detected in the PM₁₀ samples from Site 3.

Thorium-230. Thorium-230 was not detected in the PM₁₀ samples from Site 3.

Uranium. The average uranium PM₁₀ concentration at Site 3 was below the Region 10 screening level. Compared to background, the average concentration of uranium at Site 3 was greater than the background average concentration but below the single detected concentration of uranium at Site 6.

In summary, the Site 3 average concentrations of phosphorus, lead-210, and polonium-210 exceeded the Region 10 screening levels as well as the background average concentrations. However, only the average concentration of phosphorus at Site 3 also exceeded the background 95th percentile concentration.

Site 4

Table 4.7-4 summarizes the air monitoring results for Site 4 and compares the results to both Site 6, the background location and the Region 10 screening value. In Table 4.7-4 background and screening values are highlighted if they are exceeded by the average analyte concentration at Site 4.

Arsenic. The average arsenic PM₁₀ concentration at Site 4 [5.87×10^{-4} µg/m³] was slightly greater than the Region 10 screening level [5.70×10^{-4} µg/m³]. Compared to background, the average concentration of arsenic at Site 4 was higher than the background

average concentration [$4.42 \times 10^{-4} \mu\text{g}/\text{m}^3$] but less than one-half the background 95th percentile concentration [$1.45 \times 10^{-3} \mu\text{g}/\text{m}^3$].

Cadmium. The average cadmium PM_{10} concentration at Site 4 was below the Region 10 screening level. Compared to background, the average concentration of cadmium at Site 4 was higher than the background average concentration, but less than the 95th percentile of background cadmium PM_{10} concentrations.

Chromium. The average total chromium PM_{10} concentration at Site 4 was slightly greater than the Region 10 chromium (VI) screening level. Converting the average concentration of total chromium at Site 4 to the maximum percentage estimated to exist as chromium (VI), the value is $8.54 \times 10^{-6} \mu\text{g}/\text{m}^3$. This value is less than the Region 10 screening value for chromium (VI) of $2.00 \times 10^{-4} \mu\text{g}/\text{m}^3$. Compared to background, the average concentration of total chromium at Site 4 was higher than both the background average concentration and the background 95th percentile concentration.

Nickel. The average nickel PM_{10} concentration at Site 4 was below the Region 10 screening level. Compared to background, the average concentration of nickel at Site 4 was higher than the background average concentration, but less than the one detected concentration at Site 6.

Phosphorus. The average phosphorus PM_{10} concentration at Site 4 was below the Region 10 screening level. Compared to background, the average concentration of phosphorus at Site 4 was below the background average concentration and the background 95th percentile concentration.

Vanadium. The average vanadium PM_{10} concentration at Site 4 was below the Region 10 screening level. Compared to background, the average concentration of vanadium at Site 4

was greater than the background average concentration, but less than the 95th percentile background concentration.

Zinc. The average zinc PM₁₀ concentration at Site 4 was below the EPA Region III screening level. Compared to background, the average concentration of zinc at Site 4 was higher than both the background average concentration and the background 95th percentile concentration.

Lead-210. The average lead-210 PM₁₀ concentration at Site 4 [2.52×10^{-2} pCi/m³] was greater than the Region 10 screening level [1.20×10^{-3} pCi/m³]. However, the average background lead-210 concentration was also above the Region 10 screening level. Compared to background, the average concentration of lead-210 at Site 4 was slightly greater than the background average concentration [2.35×10^{-2} pCi/m³] but less than one-half the 95th percentile concentration [5.41×10^{-2} pCi/m³].

Polonium-210. The average polonium-210 PM₁₀ concentration at Site 4 [8.38×10^{-3} pCi/m³] was above the Region 10 screening level [1.80×10^{-3} pCi/m³]. However, the average background polonium-210 concentration was also above the screening level. Compared to background, the average concentration of polonium-210 at Site 4 was higher than the background average concentration [6.33×10^{-3} pCi/m³] but less than one-half the 95th percentile background concentration [2.05×10^{-2} pCi/m³].

Radium-226. Radium-226 was not detected in the PM₁₀ samples from Site 4.

Thorium-230. Thorium-230 was not detected in the PM₁₀ samples from Site 4.

Uranium. The average uranium PM₁₀ concentration at Site 4 was below the Region 10 screening level. Compared to background, the average concentration of uranium at Site 4 was greater than the background average concentration, but below the one detected uranium background concentration.

In summary, the average concentrations of arsenic, lead-210, and polonium-210 exceeded both the Region 10 screening levels and the background average concentrations at Site 4. However, no average chemical concentrations at Site 4 exceeded both the screening level and the background 95th percentile concentration.

Site 5

Table 4.7-5 summarizes the air monitoring results for Site 5 and compares the results to both Site 6, the background location, and the Region 10 screening levels. In Table 4.7-5 background and screening values are highlighted if they are exceeded by the average analyte concentration for Site 5.

Arsenic. The average arsenic PM_{10} concentration at Site 5 was less than the Region 10 screening level. Compared to background, the average concentration of arsenic at Site 5 was greater than the background average concentration but less than one-half the background 95th percentile concentration.

Cadmium. The average cadmium PM_{10} concentration at Site 5 [$1.54 \times 10^{-3} \mu\text{g}/\text{m}^3$] was 10 percent greater than the Region 10 screening level [$1.40 \times 10^{-3} \mu\text{g}/\text{m}^3$]. Compared to background, the average concentration of cadmium at Site 5 was greater than both the background average concentration [$7.00 \times 10^{-4} \mu\text{g}/\text{m}^3$] and the background 95th percentile concentration [$1.11 \times 10^{-3} \mu\text{g}/\text{m}^3$].

Chromium. The average total chromium PM_{10} concentration at Site 5 [$2.35 \times 10^{-3} \mu\text{g}/\text{m}^3$] was greater than the Region 10 chromium (VI) screening level [$2.00 \times 10^{-4} \mu\text{g}/\text{m}^3$]. Converting the average concentration of total chromium at Site 5 to the maximum percentage estimated to exist as chromium (VI), the value is $2.35 \times 10^{-5} \mu\text{g}/\text{m}^3$. This value is less than the Region 10 screening value for chromium (VI) of $2.00 \times 10^{-4} \mu\text{g}/\text{m}^3$. Compared to background, the average

concentration of total chromium at Site 5 was greater than the background average concentration [$1.34 \times 10^{-4} \mu\text{g}/\text{m}^3$] and the background 95th percentile concentration [$6.58 \times 10^{-4} \mu\text{g}/\text{m}^3$].

Nickel. The average nickel PM_{10} concentration at Site 5 was less than the Region 10 screening level. The average PM_{10} nickel concentration at Site 5 is approximately equal to the background average concentration. The average concentration of nickel at Site 5 was less than the one detected concentration at Site 6.

Phosphorus. The average phosphorus PM_{10} concentration at Site 5 [$4.53 \times 10^{-1} \mu\text{g}/\text{m}^3$] was greater than the Region 10 screening level [$3.00 \times 10^{-1} \mu\text{g}/\text{m}^3$]. Compared to background, the average concentration of phosphorus at Site 5 was greater than both the background average concentration [$3.34 \times 10^{-2} \mu\text{g}/\text{m}^3$] and the background 95th percentile concentration [$2.03 \times 10^{-1} \mu\text{g}/\text{m}^3$].

Vanadium. The average vanadium PM_{10} concentration at Site 5 was less than the Region 10 screening level. Compared to background, the average concentration of vanadium at Site 5 was greater than both the background average concentration and the background 95th percentile concentration.

Zinc. The average zinc PM_{10} concentration at Site 5 was less than the EPA Region III screening level. Compared to background, the average concentration of zinc at Site 5 was greater than the background average concentration but less than the background 95th percentile concentration.

Lead-210. The average lead-210 PM_{10} concentration at Site 5 [$2.26 \times 10^{-2} \text{pCi}/\text{m}^3$] was greater than the Region 10 screening level [$1.20 \times 10^{-3} \text{pCi}/\text{m}^3$]. However, the average background lead-210 concentration was also above the Region 10 screening level. Compared to background, the average concentration of lead-210 at Site 5 was less than both the background

average concentration [2.35×10^{-2} pCi/m³] and the background 95th percentile concentration [5.41×10^{-2} pCi/m³].

Polonium-210. The average polonium-210 PM₁₀ concentration at Site 5 [1.47×10^{-2} pCi/m³] was greater than the Region 10 screening level [1.80×10^{-3} pCi/m³]. However, the average background polonium-210 concentration is also above the Region 10 screening level. Compared to background, the average concentration of polonium-210 at Site 5 was greater than the background average concentration [6.33×10^{-3} pCi/m³] but less than the background 95th percentile concentration [2.05×10^{-2} pCi/m³].

Radium-226. Radium-226 was not detected in the PM₁₀ samples from Site 5.

Thorium-230. The average thorium-230 PM₁₀ concentration at Site 5 was less than the Region 10 screening level. Compared to background, the average concentration of thorium at Site 5 was higher the background average concentration. (Thorium-230 was not detected in samples from Site 6.)

Uranium. The average uranium PM₁₀ concentration at Site 5 was less than the Region 10 screening level. Compared to background, the average concentration of uranium at Site 5 is greater than the background average concentration but less than the one detected background concentration.

In summary, the Site 5 average concentrations of cadmium, phosphorus, and polonium-210 exceeded both the Region 10 screening levels and the background average concentrations. However, only the average concentrations of cadmium and phosphorus at Site 5 were also greater than the background 95th percentile concentrations.

Perimeter Sites

The air monitoring stations at Sites 1, 2, and 7 were sited to represent the likely maximally impacted areas. However, only Sites 3, 4, and 5 represent potential human exposure points and

only these sites are appropriate for comparison to the ambient air screening levels. Residents currently do not live in the vicinity of Sites 1, 2, and 7. This is also expected to be the case in the future because the area borders the plant areas, is mostly company-owned, and in many cases is or will be deed restricted to prevent non-industrial use.

A summary of the data from the perimeter Sites (1, 2, and 7) is presented in Tables 4.7-6 through 4.7-8. A detailed evaluation of the data from the perimeter sites, as well as all sample results, is provided in Part III, Volume 1. The related air modeling studies, for which data from these perimeter sites were collected, are presented in Part III, Volume 2.

4.7.4 SUMMARY OF FINDINGS

The PM_{10} fraction of particulate matter is the most relevant to potential human health effects resulting from air exposure. The concern at the EMF site is the potential effects of chronic exposure to low concentrations of particles and constituents in air. Therefore, average values of PM_{10} and its constituents were calculated from the 24-hour sample data. Sites 3, 4, and 5 are located near human populations in residential areas. The average concentrations from these locations were compared to the values for Site 6, the upwind background location. Air concentrations of naturally occurring constituents often exhibit a great deal of variability (EPA, 1989). Therefore, the 95th percentile of background concentrations, representing the upper end of the background range of concentrations, was also used for comparisons.

The community Sites 3, 4, and 5 showed very similar average values of PM_{10} mass, within the range measured at the background site. Consequently, the measurements made at the community sites can be considered to represent ambient PM_{10} mass levels. The daily PM_{10} mass concentrations for Sites 3, 4, and 5 all correlated well with Site 6, the background site. This strong correlation indicates that the residential sites are being primarily influenced by the same factors that affect background.

Comparison of Sites 3, 4, and 5 samples with Site 6 showed most PM₁₀ constituents were present at concentrations below risk-based screening levels or at background concentrations. It should be noted that the average background concentrations of chromium, lead-210, and polonium-210 also exceed the Region 10 air screening levels. Although there was elevation of some monitored PM₁₀ constituents over both average background and screening levels, in general, the average air concentrations of constituents present in concentrations greater than the background average concentrations and screening levels were less than 2-fold greater than the average background concentrations, and less than the 95th percentile background concentrations. The exceptions are phosphorus at Sites 3 and 5, and cadmium at Site 5.

Surface and Subsurface Characterizations
Tables for Section 4.7

Section 4 Nature and Extent of Constituents of Potential Concern

TABLE 4.7-1
MASS PM₁₀

Sampling Station	Detection Frequency	Range of Detected Values (µg/m ³)	Arithmetic Mean (µg/m ³)
Background			
6	195 / 195	2.10E-01 - 1.06E+02	1.94E+01
Community			
3	142 / 142	1.49E+00 - 5.30E+01	1.99E+01
4	178 / 178	2.12E+00 - 6.92E+01	2.34E+01
5	175 / 175	2.00E-01 - 9.08E+01	1.91E+01
Perimeter			
1	190 / 190	4.11E+00 - 7.95E+01	3.01E+01
2 ^(a)	191 / 191	6.56E+00 - 1.51E+02	5.67E+01
7	186 / 186	6.00E-01 - 1.19E+02	2.05E+01

Note: (a) Site 2 does not comply with EPA's Monitoring Guidelines (EPA-450/4-87-007, May 1987) and thus does not represent "ambient air."

TABLE 4.7-2
PM₁₀ CONSTITUENTS
BACKGROUND CONCENTRATIONS

Station 6					Ambient Air Screening Levels ^(a)
Analyte	Detection Frequency	Range of Detected Values	Arithmetic Mean	95th Percentile	
METALS ($\mu\text{g}/\text{m}^3$)					
Arsenic	20 / 46	1.69E-04 - 3.75E-03	4.42E-04	1.45E-03	5.70E-04
Cadmium	2 / 46	1.35E-03 - 1.61E-03	7.00E-04	1.11E-03	1.40E-03
Chromium (Total)	4 / 46	5.70E-04 - 7.70E-04	1.34E-04	6.58E-04	—
Chromium VI					2.00E-04
Nickel	1 / 36	3.20E-03	1.71E-03	3.20E-03 M	1.00E-02
Phosphorus	3 / 46	1.88E-01 - 7.78E-01	3.34E-02	2.03E-01	3.00E-01
Vanadium	3 / 46	1.61E-03 - 1.95E-03	8.96E-04	1.71E-03	1.70E-01
Zinc	34 / 46	1.92E-03 - 1.20E-02	4.67E-03	1.13E-02	1.10E+03 ^(b)
RADIONUCLIDES (pCi/m³)					
Lead-210	43 / 45	3.29E-03 - 7.66E-02	2.35E-02	5.41E-02	1.20E-03
Polonium-210	39 / 46	2.05E-03 - 3.93E-02	6.33E-03	2.05E-02	1.80E-03
Radium-226	1 / 46	3.33E-03	3.32E-04	3.33E-03 M	1.60E-03
Thorium-230	0 / 46		1.75E-05 U	1.75E-05 U	2.00E-04
Uranium	1 / 46	4.41E-04	1.12E-05	4.41E-04 M	1.00E-04

U = all nondetected values

M = only one detected concentration, detected value shown

Highlighting indicates that the value exceeds the Region 10 Proposed Ambient Air Screening Levels for the constituent.

Notes: (a) The Ambient Air Screening Levels presented are from Table 4-1, July 2, 1993 letter from W. Adams (EPA Region 10) to R. Hosking (FMC) and E. Mapes (Simplot).

(b) The Ambient Air Screening Level for zinc was not determined by Region 10; therefore, the Region 3 risk-based concentration for zinc is presented, 1.1E+03 $\mu\text{g}/\text{m}^3$ (EPA, 1995a).

Section 4 Nature and Extent of Constituents of Potential Concern

TABLE 4.7-3
PM₁₀ CONSTITUENTS
SAMPLING STATION 3

Station 3				Background (Station 6)		Ambient Air Screening Levels ^(a)
Analyte	Detection Frequency	Range of Detected Values	Arithmetic Mean	Arithmetic Mean	95th Percentile	
METALS (µg/m ³)						
Arsenic	35 / 43	1.72E-04 - 3.39E-03	5.70E-04	4.42E-04	1.45E-03	5.70E-04
Cadmium	17 / 43	1.34E-03 - 3.63E-03	1.23E-03	7.00E-04	1.11E-03	1.40E-03
Chromium (Total)	15 / 43	1.07E-03 - 4.97E-03	9.24E-04	1.34E-04	6.58E-04	---
Chromium VI			9.24E-06			2.00E-04
Nickel	4 / 35	3.47E-03 - 3.70E-03	1.89E-03	1.71E-03	3.20E-03 M	1.00E-02
Phosphorus	13 / 43	2.68E-01 - 1.53E+00	3.22E-01	3.34E-02	2.03E-01	3.00E-01
Vanadium	12 / 43	1.80E-03 - 4.49E-03	1.39E-03	8.96E-04	1.71E-03	1.70E-01
Zinc	40 / 43	6.33E-03 - 2.40E-02	1.35E-02	4.67E-03	1.13E-02	1.10E+03 ^(b)
RADIONUCLIDES (pCi/m ³)						
Lead-210	41 / 42	2.28E-03 - 8.37E-02	2.51E-02	2.35E-02	5.41E-02	1.20E-03
Polonium-210	41 / 42	2.32E-03 - 3.82E-02	1.25E-02	6.33E-03	2.05E-02	1.80E-03
Radium-226	0 / 42		2.65E-04 U	3.32E-04	3.33E-03 M	1.60E-03
Thorium-230	0 / 41		1.75E-05 U	1.75E-05 U	1.75E-05 U	2.00E-04
Uranium	5 / 42	1.52E-04 - 2.28E-04	2.38E-05	1.12E-05	4.41E-04 M	1.00E-04

U = all nondetected values

M = only one detected concentration, detected value shown

Highlighting indicates that the value is exceeded by the arithmetic mean measured at the site.

Notes: (a) The Ambient Air Screening Levels presented are from Table 4-1, July 2, 1993 letter from W. Adams (EPA Region 10) to R. Hosking (FMC) and E. Mapes (Simplot).

(b) The Ambient Air Screening Level for zinc was not determined by Region 10; therefore, the Region 3 risk-based concentration for zinc is presented, 1.1E+03 $\mu\text{g}/\text{m}^3$ (EPA, 1995a).

Section 4 Nature and Extent of Constituents of Potential Concern

TABLE 4.7-4
PM₁₀ CONSTITUENTS
SAMPLING STATION 4

Station 4				Background (Station 6)		Ambient Air Screening Levels ^(a)
Analyte	Detection Frequency	Range of Detected Values	Arithmetic Mean	Arithmetic Mean	95th Percentile	
METALS (µg/m ³)						
Arsenic	33 / 45	1.92E-04 - 3.00E-03	5.87E-04	4.42E-04	1.45E-03	5.70E-04
Cadmium	6 / 45	1.32E-03 - 1.90E-03	7.93E-04	7.00E-04	1.11E-03	1.40E-03
Chromium (Total)	16 / 45	1.27E-03 - 7.04E-03	8.54E-04	1.34E-04	6.58E-04	---
Chromium VI			8.54E-06			2.00E-04
Nickel	3 / 36	3.17E-03 - 3.81E-03	1.81E-03	1.71E-03	3.20E-03 M	1.00E-02
Phosphorus	2 / 45	2.05E-01 - 3.39E-01	2.01E-02	3.34E-02	2.03E-01	3.00E-01
Vanadium	11 / 45	1.72E-03 - 2.49E-03	1.14E-03	8.96E-04	1.71E-03	1.70E-01
Zinc	42 / 45	3.44E-03 - 2.86E-02	1.22E-02	4.67E-03	1.13E-02	1.10E+03 ^(b)
RADIONUCLIDES (pCi/m ³)						
Lead-210	44 / 44	2.31E-03 - 8.07E-02	2.52E-02	2.35E-02	5.41E-02	1.20E-03
Polonium-210	42 / 44	2.38E-03 - 3.64E-02	8.38E-03	6.33E-03	2.05E-02	1.80E-03
Radium-226	0 / 44		2.65E-04 U	3.32E-04	3.33E-03 M	1.60E-03
Thorium-230	0 / 44		1.75E-05 U	1.75E-05 U	1.75E-05 U	2.00E-04
Uranium	3 / 44	1.40E-04 - 2.03E-04	1.32E-05	1.12E-05	4.41E-04 M	1.00E-04

U = all nondetected values

M = only one detected concentration, detected value shown

Highlighting indicates that the value is exceeded by the arithmetic mean measured at the site.

- Notes:** (a) The Ambient Air Screening Levels presented are from Table 4-1, July 2, 1993 letter from W. Adams (EPA Region 10) to R. Hosking (FMC) and E. Mapes (Simplot).
(b) The Ambient Air Screening Level for zinc was not determined by Region 10; therefore, the Region 3 risk-based concentration for zinc is presented, 1.1E+03 µg/m³ (EPA,1995a).

TABLE 4.7-5
PM₁₀ CONSTITUENTS
SAMPLING STATION 5

Station 5				Background (Station 6)		Ambient Air Screening Levels ^(a)
Analyte	Detection Frequency	Range of Detected Values	Arithmetic Mean	Arithmetic Mean	95th Percentile	
METALS (µg/m ³)						
Arsenic	28 / 43	1.85E-04 - 2.36E-03	5.50E-04	4.42E-04	1.45E-03	5.70E-04
Cadmium	13 / 43	1.34E-03 - 8.26E-03	1.54E-03	7.00E-04	1.11E-03	1.40E-03
Chromium (Total)	14 / 43	8.00E-04 - 6.00E-02	2.35E-03	1.34E-04	6.58E-04	—
Chromium VI			2.35E-05			2.00E-04
Nickel	1 / 34	3.65E-03	1.72E-03	1.71E-03	3.20E-03 M	1.00E-02
Phosphorus	15 / 43	4.49E-01 - 2.08E+00	4.53E-01	3.34E-02	2.03E-01	3.00E-01
Vanadium	13 / 43	1.69E-03 - 6.13E-02	2.77E-03	8.96E-04	1.71E-03	1.70E-01
Zinc	41 / 43	1.16E-03 - 4.24E-02	9.87E-03	4.67E-03	1.13E-02	1.10E+03 ^(b)
RADIONUCLIDES (pCi/m ³)						
Lead-210	42 / 43	2.59E-03 - 6.25E-02	2.26E-02	2.35E-02	5.41E-02	1.20E-03
Polonium-210	42 / 43	1.98E-03 - 9.94E-02	1.47E-02	6.33E-03	2.05E-02	1.80E-03
Radium-226	0 / 43		2.65E-04 U	3.32E-04	3.33E-03 M	1.60E-03
Thorium-230	2 / 43	7.22E-04 - 9.24E-04	5.50E-05	1.75E-05 U	1.75E-05 U	2.00E-04
Uranium	8 / 43	1.57E-04 - 4.74E-04	4.72E-05	1.12E-05	4.41E-04 M	1.00E-04

U = all nondetected values

M = only one detected concentration, detected value shown

Highlighting indicates that the value is exceeded by the arithmetic mean measured at the site.

Notes: (a) The Ambient Air Screening Levels presented are from Table 4-1, July 2, 1993 letter from W. Adams (EPA Region 10) to R. Hosking (FMC) and E. Mapes (Simplot).

(b) The Ambient Air Screening Level for zinc was not determined by Region 10; therefore, the Region 3 risk-based concentration for zinc is presented, 1.1E+03 $\mu\text{g}/\text{m}^3$ (EPA, 1995a).

Section 4 Nature and Extent of Constituents of Potential Concern

TABLE 4.7-6
PM₁₀ CONSTITUENTS
SAMPLING STATION 1

Station 1				Background (Station 6)	
Analyte	Detection Frequency	Range of Detected Values	Arithmetic Mean	Arithmetic Mean	95th Percentile
METALS (µg/m³)					
Arsenic	36 / 46	2.20E-04 - 2.77E-03	6.23E-04	4.42E-04	1.45E-03
Cadmium	24 / 46	9.78E-04 - 8.87E-03	2.13E-03	7.00E-04	1.11E-03
Chromium (Total)	30 / 46	1.21E-03 - 4.02E-02	4.02E-03	1.34E-04	6.58E-04
Chromium VI			4.02E-05		
Nickel	5 / 36	2.42E-03 - 6.61E-03	2.13E-03	1.71E-03	3.20E-03 M
Phosphorus	24 / 46	7.66E-01 - 6.22E+00	8.81E-01	3.34E-02	2.03E-01
Vanadium	28 / 46	1.84E-03 - 4.28E-02	4.29E-03	8.96E-04	1.71E-03
Zinc	45 / 46	6.56E-03 - 1.19E-01	2.00E-02	4.67E-03	1.13E-02
RADIONUCLIDES (pCi/m³)					
Lead-210	44 / 46	2.85E-03 - 1.17E-01	2.49E-02	2.35E-02	5.41E-02
Polonium-210	44 / 46	2.69E-03 - 4.84E-02	1.53E-02	6.33E-03	2.05E-02
Radium-226	0 / 46		2.65E-04 U	3.32E-04	3.33E-03 M
Thorium-230	5 / 45	1.97E-04 - 8.02E-04	7.94E-05	1.75E-05 U	1.75E-05 U
Uranium	24 / 46	1.39E-04 - 2.24E-03	1.89E-04	1.12E-05	4.41E-04 M

U = all nondetected values

M = only one detected concentration, detected value shown

Highlighting indicates that the value is exceeded by the arithmetic mean measured at the site.

Note: Site 1 was located for purposes of model validation and is not representative of residential exposure.

Section 4 Nature and Extent of Constituents of Potential Concern

TABLE 4.7-7
PM₁₀ CONSTITUENTS
SAMPLING STATION 2

Station 2				Background (Station 6)	
Analyte	Detection Frequency	Range of Detected Values	Arithmetic Mean	Arithmetic Mean	95th Percentile
METALS ($\mu\text{g}/\text{m}^3$)					
Arsenic	44 / 45	1.80E-04 - 4.61E-03	1.29E-03	4.42E-04	1.45E-03
Cadmium	40 / 45	1.38E-03 - 5.60E-02	1.17E-02	7.00E-04	1.11E-03
Chromium (Total)	45 / 45	2.23E-03 - 1.19E-01	1.76E-02	1.34E-04	6.58E-04
Chromium VI			1.76E-04		
Nickel	19 / 36	3.23E-03 - 2.25E-02	4.15E-03	1.71E-03	3.20E-03 M
Phosphorus	44 / 45	7.45E-01 - 1.91E+01	5.53E+00	3.34E-02	2.03E-01
Vanadium	44 / 45	1.63E-03 - 1.26E-01	1.96E-02	8.96E-04	1.71E-03
Zinc	45 / 45	1.50E-02 - 4.16E-01	8.07E-02	4.67E-03	1.13E-02
RADIONUCLIDES (pCi/m^3)					
Lead-210	42 / 45	5.97E-03 - 7.46E-02	2.44E-02	2.35E-02	5.41E-02
Polonium-210	42 / 43	2.28E-03 - 3.51E-01	7.31E-02	6.33E-03	2.05E-02
Radium-226	1 / 45	8.48E-04	2.78E-04	3.32E-04	3.33E-03 M
Thorium-230	25 / 38	2.01E-04 - 1.50E-03	2.94E-04	1.75E-05 U	1.75E-05 U
Uranium	41 / 45	1.95E-04 - 5.29E-03	8.01E-04	1.12E-05	4.41E-04 M

U = all nondetected values

M = only one detected concentration, detected value shown

Note: Highlighting indicates that the value is exceeded by the arithmetic mean measured at the site.

Note: Site 2 was located for purposes of model validation and is not representative of residential exposure. It does not comply with EPA's Monitoring Guidelines (EPA-450/4-87-007, May 1987) or with EPA's Guidelines for Exposure Assessment, 57 Federal Register 22888 (May 29, 1992). The data presented here thus do not represent "ambient air" 40 C.F.R. Section 50.1(e), worker exposure, or residential exposure.

TABLE 4.7-8
PM₁₀ CONSTITUENTS
SAMPLING STATION 7

Station 7				Background (Station 6)	
Analyte	Detection Frequency	Range of Detected Values	Arithmetic Mean	Arithmetic Mean	95th Percentile
METALS ($\mu\text{g}/\text{m}^3$)					
Arsenic	24 / 44	1.67E-04 - 2.06E-03	4.80E-04	4.42E-04	1.45E-03
Cadmium	17 / 44	1.36E-03 - 9.88E-03	1.48E-03	7.00E-04	1.11E-03
Chromium (Total)	15 / 44	7.90E-04 - 4.40E-03	8.24E-04	1.34E-04	6.58E-04
Chromium VI			8.24E-06		
Nickel	1 / 35	3.68E-03	1.72E-03	1.71E-03	3.20E-03 M
Phosphorus	14 / 44	7.14E-01 - 1.97E+00	3.43E-01	3.34E-02	2.03E-01
Vanadium	13 / 44	1.67E-03 - 5.60E-03	1.36E-03	8.96E-04	1.71E-03
Zinc	35 / 44	2.40E-03 - 4.08E-02	8.28E-03	4.67E-03	1.13E-02
RADIONUCLIDES (pCi/m^3)					
Lead-210	42 / 44	2.20E-03 - 6.26E-02	2.17E-02	2.35E-02	5.41E-02
Polonium-210	41 / 44	2.00E-03 - 5.55E-02	1.30E-02	6.33E-03	2.05E-02
Radium-226	0 / 44		2.65E-04 U	3.32E-04	3.33E-03 M
Thorium-230	0 / 40		1.75E-05 U	1.75E-05 U	1.75E-05 U
Uranium	6 / 44	1.45E-04 - 2.46E-04	2.58E-05	1.12E-05	4.41E-04 M

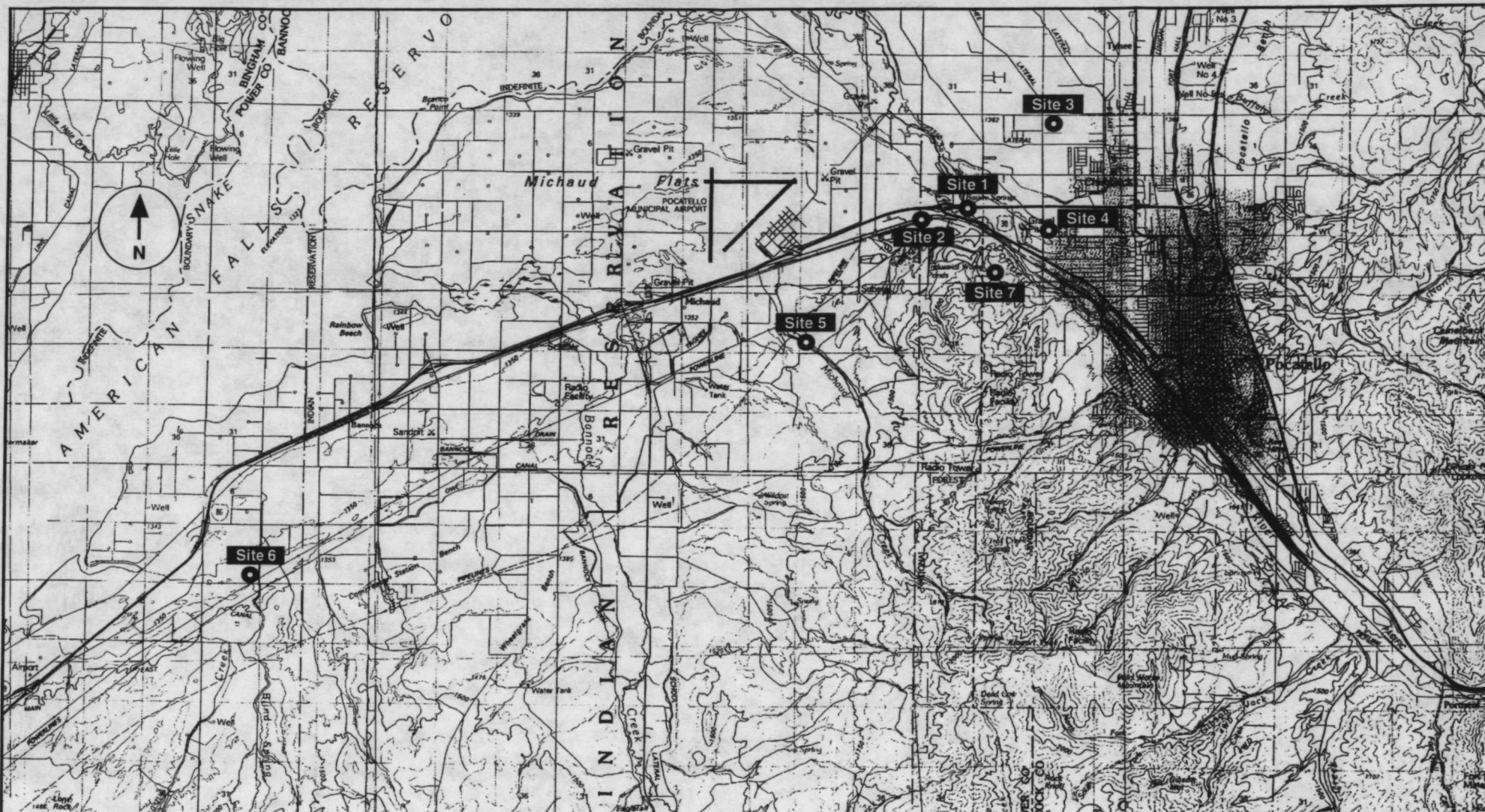
U = all nondetected values

M = only one detected concentration, detected value shown

Note: Highlighting indicates that the value is exceeded by the arithmetic mean measured at the site.

Note: Site 7 was located for purposes of model validation and is not representative of residential exposure.

Surface and Subsurface Characterizations
Figures for Section 4.7



Legend:

● - Monitoring Site

0 1 2 3 4 5 miles

BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

EMF Air Monitoring Sites



JOB NO.
21372

DRAWING NO.
FIGURE 4.7-1

REV.

Fate and Transport

This section describes the environmental fate and transport of chemicals originating from the EMF facilities. It focuses on two environmental transport pathways associated with the EMF facilities – the unsaturated soils or vadose zone beneath the facilities and the groundwater system beneath and downgradient of the facilities. A third pathway, air, is addressed in RI Report Part III.

This section is comprised of five major parts. Section 5.1 describes the principal routes of chemical migration from the EMF site that emerge from the assessment of the physical and chemical site characterization data presented in Sections 3 and 4. Section 5.2 addresses the general persistence of chemicals originating from the EMF facilities. Section 5.3 describes the factors affecting the migration of EMF-related constituents in the subsurface environment (i.e., vadose zone and groundwater). Section 5.4 presents a solute flux analysis that quantifies the flux of EMF-related constituents in groundwater, as well as the influence of EMF-affected groundwater on the Portneuf River relative to other natural and anthropogenic sources. Section 5.5 summarizes the discussions provided in Section 5.1 through 5.4.

5.1 PRINCIPAL ROUTES OF MIGRATION AND SITE CONCEPTUAL MODEL

This section describes the principal release mechanisms and pathways by which EMF-related constituents move through the environment. A schematic representation of these mechanisms and pathways is presented in the site conceptual model, Figure 5.1-1. The assessment of impact on potential receptor populations, if any, is beyond the scope of this section and has been addressed in a separate EMF risk assessment prepared by EPA (E&E, 1995).

The conceptual model is a synthesis of the physical and site characterization data presented in Sections 3 and 4. A discussion of the conceptual model, organized by transport pathway or the media to which a receptor is potentially exposed (e.g., air, soil, groundwater), is provided below.

Air. The EMF facilities emit a number of constituents as described by the facility emissions inventories presented in RI Report Part III, Volume 2. Human receptors may be exposed to airborne chemicals via inhalation.

Surface and Near-Surface Soils. Mixing of soils with facility products, byproducts and other fill materials, and facility spills or other releases have affected surface and/or near-surface soils within portions of the industrial operations and waste management areas of the EMF facilities, as indicated by the facility soil data summarized in Section 4.2, Potential Sources and Facility Soils. EMF facility air emissions have affected surface soils in portions of the EMF study area, as described in Section 4.3, Surface Soils. The principal areas of affected soils are within the property boundaries of the facilities, and land use controls restrict human exposure in these areas.

Plants. As described in Section 4.6, Ecology, chemicals in surface soils can be taken up by plants which in turn may be ingested by human and/or other terrestrial receptors.

Vadose Zone. As described in Section 4.2, at some facility sources EMF chemicals have migrated through surface or near-surface facility soils into deeper soils via infiltration and deep percolation of aqueous byproducts and/or wastes.

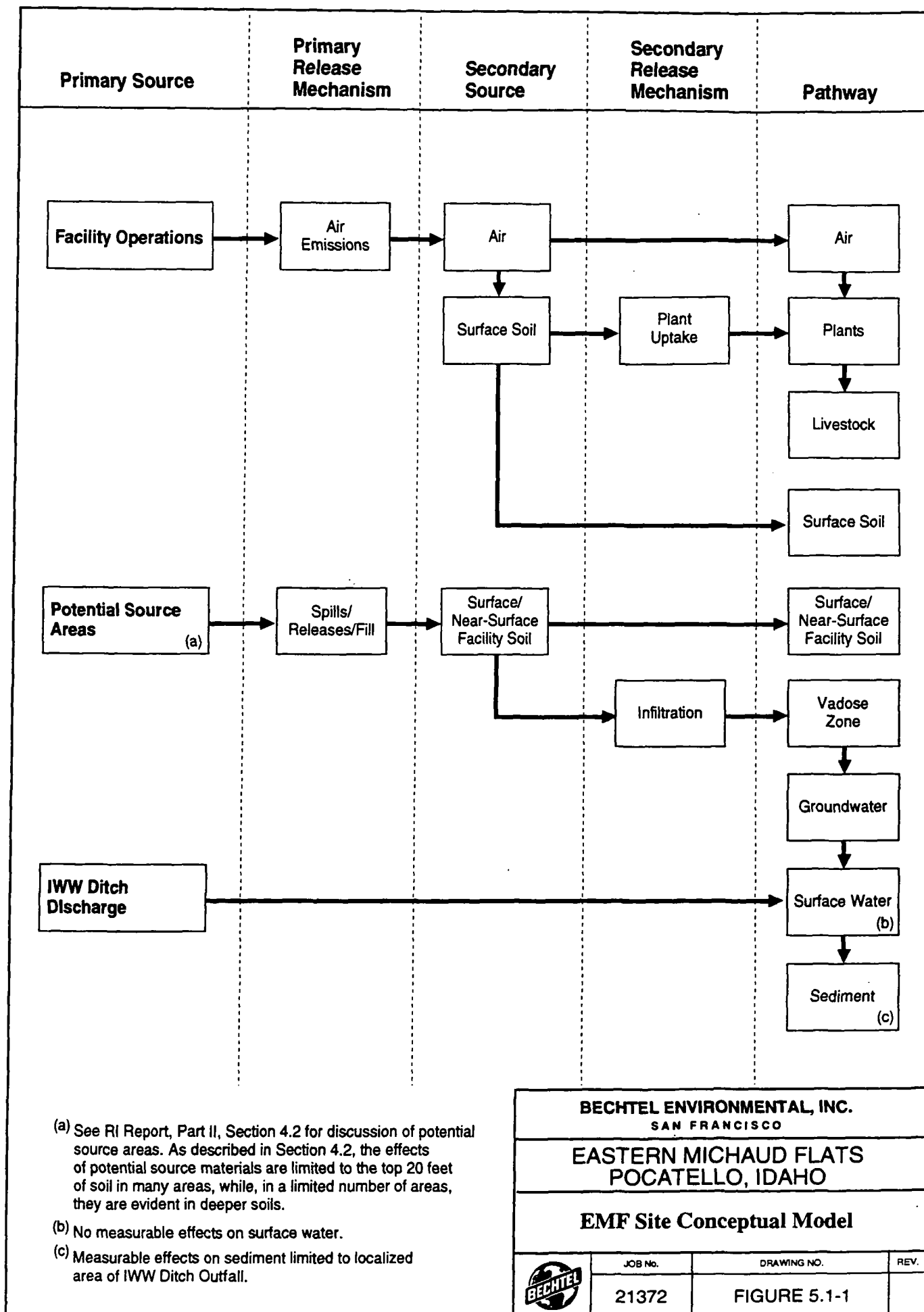
Groundwater. As indicated in Section 4.4, Groundwater, some EMF chemicals have migrated through unsaturated soils to groundwater. However, groundwater affected by chemicals originating from the EMF facilities is not a current source of drinking water.

Surface Water/Sediment. As described in Section 4.5, Surface Water and Sediments, EMF-affected groundwater ultimately discharges to the Portneuf River via springs and base flow contribution. Nevertheless, with the possible exception of sulfate, the effect of this discharge on surface water quality is not measurable in the river. Likewise, the effects of discharges to the river via the IWW ditch outfall are not measurable in surface water, or in sediment beyond the localized outfall area. Cadmium concentrations in the Portneuf River delta were elevated in comparison to upstream Portneuf River and Snake River delta sediment samples.

As described in Section 3.2, Drainage and Surface Water Hydrology, surface water runoff generated at the facilities is retained within the facility boundaries and does not discharge to the Portneuf River. Furthermore, as described in Section 4.5, Surface Water and Sediment, the effects, if any, of surface water runoff from EMF-affected surface soils outside the facility boundaries were not measurable in the river. Consequently, surface water runoff is not considered a significant release mechanism for chemicals originating from the EMF facilities and is not indicated on the conceptual model shown in Figure 5.1-1.

In summary, the primary pathways (or media) by which chemicals originating from the EMF facilities are transported in the environment are air, surface and near-surface soils, the vadose zone and groundwater. Chemical fate and transport via the air pathway is addressed in RI Report Part III. The fate and transport of chemicals in the remaining pathways are the subject of Sections 5.3 and 5.4.

Surface and Subsurface Characterizations
Figures for Section 5.1



5.2 PERSISTENCE OF CONSTITUENTS OF CONCERN

Chemicals originating from the EMF facilities are principally derived from phosphate ore, the primary feedstock for both the FMC and Simplot plants. The ore contains apatite, a mineral containing calcium, phosphorus and fluoride. The ore also contains trace amounts of arsenic, cadmium, chromium, vanadium, uranium-238 (and its radioactive daughters) and other elements. These constituents, given their inorganic nature, are persistent in the environment, unlike many organic compounds which degrade over time.

5.3 FACTORS AFFECTING SUBSURFACE CONSTITUENT MIGRATION

This section describes the factors affecting the migration of inorganic constituents from EMF facility sources along flowpaths through the vadose zone and groundwater. The fate and transport of chemicals in the subsurface is potentially affected by a variety of chemical, biological and physical reactions, transformations and processes which may take place in the unsaturated and saturated subsurface environment. These include the following factors:

- seepage rates
- pH change and soil mineralogy
- aquifer geochemistry
- advective mixing

Many of the chemicals originating from the EMF facilities decline in concentration to below representative levels along one or more of the principal migration pathways. The decreases in chemical concentrations can be explained by one or more attenuation factors including ionic exchange and precipitation, as well as by advective mixing. The chemicals can be loosely grouped into those that are highly attenuated in the vadose zone (e.g., cadmium), those that are rapidly attenuated upon introduction to groundwater (e.g., zinc), those that are attenuated along the groundwater flowpath (e.g., fluoride), and those that are relatively unattenuated in groundwater (e.g., sulfate).

Site hydrogeology, climate, and the facility soil and groundwater data summarized in Sections 3.3, 3.5, 4.2, and 4.4 indicate that the EMF-related constituents in groundwater have originated at source areas to which a sustained, artificial head, containing high solute concentrations, is or was applied. These sources areas include the former unlined ponds and the gypsum stacks. The principal factors affecting constituent migration in the vadose zone beneath the EMF facilities are seepage rates and seepage fluid and soil chemistry. The principal factors affecting migration in the saturated zone are attenuation and advective mixing.

This section is devoted to discussion of the subsurface migration factors identified above. Factors affecting vadose zone migration, namely seepage rates and fluid and soil chemistry, are discussed in Section 5.3.1. Factors affecting constituent migration in groundwater, namely attenuation and advective mixing, are discussed in Section 5.3.2. The discussion relies on empirical observations based on the RI data with support from the literature and various prior studies of EMF source seepage and/or geochemistry.

5.3.1 PRINCIPAL FACTORS AFFECTING MIGRATION IN THE VADOSE ZONE

This section begins with a discussion of seepage rates from the former unlined ponds and the gypsum stacks and the factors that affect seepage rates from these sources (Section 5.3.1.1). Section 5.3.1.2 discusses the effects of increasing soil pH on the mobility of constituents as they enter and move through the vadose zone.

5.3.1.1 Seepage Rates

As indicated in Section 4.2, Potential Source and Facility Soils, the former unlined ponds and the gypsum stacks are areas to which a sustained hydraulic head is, or was, applied. In the absence of an artificially applied head, seepage is limited to that from natural precipitation. As discussed in Section 3.3, the average infiltration throughout the Eastern Snake River Plain is about 10% of mean annual precipitation (Wood and Low, 1986), resulting in approximately 1 inch of infiltration per year.

Since 1975, when FMC began closing and filling unlined ponds, EMF facility pond seepage has been significantly reduced. Beginning in the late 1980's, Simplot initiated changes in its operation of the gypsum stacks, which further reduced seepage. Other, more recent changes at both the Simplot and FMC facilities that have reduced seepage include Simplot's closure of the former east overflow pond in 1993, and FMC's lining of the railroad swale, filling of Pond 8S, and cessation of the slag pit sump operations in 1994. The gypsum stacks are the only remaining sources within the EMF facilities with a sustained applied head.

Seepage at FMC

The surface area of an impoundment is one of the most significant factors relating to the flux of water through the vadose zone; depth of the fluids is of secondary importance (Johnson and Finlayson, 1989). The results of a water balance study at Pond 8S performed in 1992 indicated there was an average seepage rate of 106 inches per year from this then-active pond (Bechtel, 1993g). This represented a deep recharge rate of 15 gallons per minute over the entire area of the pond (117,474 square feet). After closure of this pond, there is expected to be an estimated 100-fold reduction in seepage. Assuming other old ponds at FMC had seepage rates similar to Pond 8S, the total reduction in pond seepage is estimated at approximately 320 gpm since pond closures began.

Other FMC ponds can be assumed similar to Pond 8S because of the similarity of the sludges at the base of these ponds. The pond sludges, which are an important seepage-controlling factor, are very fine-grained materials slurried to the ponds for storage. Sludges settle to the bottom of the ponds, forming a low-permeability layer that reduces seepage. Permeability reduction by settling out of suspended fine-grained sediments is a well-documented process at artificial recharge ponds (Johnson and Finlayson, 1989). Since all unlined ponds at FMC have had the head removed, their contribution to the future migration of EMF-related constituents will be greatly reduced.

Seepage at Simplot

The supernatant associated with the gypsum slurry applied to the Simplot gypsum stacks is (1) captured by the gypsum stack decant system, (2) consumed in the hydration of gypsum as excess gypsum precipitates out of the supernatant, (3) evaporated, or (4) lost as seepage. Several factors help reduce the amount lost as seepage, including the low permeability and overall thickness of the gypsum material and the rotation of slurry application areas (i.e., slurry is applied over only a portion of the stack at any one time, reducing the area and depth of ponded water and increasing the evaporation potential). Another factor that may reduce seepage is the

permeability reduction in the native soils underlying the gypsum stacks as minerals precipitate from the percolating water and cement the native soils (Cochrane, 1979).

Evaporation from the gypsum stacks may also be enhanced by the current slurry application cycle. From 1966 to 1988, Simplot applied slurry to the upper gypsum stack only, building it up to current levels. During typical operations, slurry was applied to one-half of the upper or southernmost stack for two years, then the slurry was applied to the other half for two years. In 1988, Simplot began switching the slurry application on a six-month cycle. This effectively increased the potential evaporative surface on an annual basis, reducing the amount of supernatant recharging the aquifer. It also reduced the duration of standing water (applied head) over any one part of the gypsum stack, further reducing seepage. Although Simplot estimates 500 gallons per minute are lost to the aquifer, this is an average, and reflects data collected when slurry was applied on a two-year cycle. Groundwater flow modeling (Appendix K) indicates that the average seepage rate may actually be much lower.

Although a variety of constituents were detected in the groundwater beneath the former east overflow pond during the RI period of investigation, the very small surface area of this pond effectively limited the seepage rate to the aquifer. In addition, the storage of low-pH fluid in this pond was intermittent, which also limited the amount of chemicals that ultimately reached the saturated zone.

5.3.1.2 pH Change and Soil Mineralogy

As indicated in Section 4.2, Potential Source and Facility Soils, most of the metals migrating in the soils appear to be associated with low pH fluids of relatively high ionic strength. (There was little to no evidence of metals leaching from relatively insoluble fill materials that also contained relatively high metal concentrations.) When the low-pH fluids entered the vadose zone, there was a distinct increase in pH as the fluids reacted with the alkaline carbonate-rich loess. Increasing fluid pH can result in precipitation of some metals. This pH-induced precipitation is

the most efficient mechanism to reduce concentrations of some metals in water (Schwartz and Domenico, 1992). The increase in pH within the vadose zone beneath the EMF facilities was sufficient to precipitate several metals. Such evidence of precipitation of a mineral phase may be seen in the RI data as very steep concentration gradients of particular metals in the soils beneath sources. For example, soil concentrations of chromium, cadmium and vanadium decrease rapidly with depth and increasing pH as shown in Figures 5.3-1a and 1b. Concentrations of these metals generally drop below representative levels within 10 feet of the native soil/fill or waste interface.

At the former east overflow pond, where a substantial thickness of loess was not encountered, increasing pH and corresponding decreases in metal concentrations in the soil profile were not observed. The east overflow pond was underlain by quartzitic gravel from the base of the pond to the saturated zone. Furthermore, this pond was not a sludge settling pond or storage unit, and did not have a thick layer of low permeability sludge at its base. The lack of carbonate-rich loess underlying this pond and the lack of a permeability-reducing substrate contributed to the relative mobility of metals in this area. It was at this pond that cadmium, chromium, nickel, and other metals were detected in groundwater (Section 4.4).

Adsorption onto soil grains also appears to play a role in controlling the mobility of various metal constituents in the vadose zone. The occurrence of elevated concentrations of zinc in deeper soil samples, and of cobalt in some monitoring wells, indicates these two metals are more mobile through the vadose zone than cadmium, chromium and vanadium. Zinc and cobalt, along with a number of other elements including cadmium, adsorb to calcite grains, one of the carbonate minerals in the loess.

5.3.2 PRINCIPAL FACTORS AFFECTING MIGRATION IN THE SATURATED ZONE

Constituent migration in the saturated zone can be thought of as an extension of the flowpath from the source through the vadose zone and into the aquifer. Once the percolating fluids reach

the saturated zone, aquifer geochemistry and advective mixing are the factors affecting chemical concentrations.

5.3.2.1 Aquifer Geochemistry

When low pH seepage and neutral-to-basic pH groundwater mix in the saturated zone, there is an overall increase in pH and constituents still in solution may precipitate. This pH change and subsequent decrease in metal concentrations are best illustrated with data related to the former east overflow pond. Fluids in the former east overflow pond contained elevated levels of cadmium, chromium and nickel and had a low pH (Section 4.4). Groundwater samples collected several hundred feet along the flowpath from this pond did not contain elevated levels of cadmium, chromium, or nickel, and had a significantly higher pH (Figure 5.3-2). This indicates that as pH increases along the flowpath, these metals were rapidly attenuated. Aquifer geochemistry was also a controlling factor in the decrease of all metal concentrations upon closure of the pond in August 1993 (Table 4.4-10). As illustrated on Figure 5.3-2, there is an inverse relationship between pH and solute concentrations in other areas as well (e.g., Pond 8S, old calciner ponds, and gypsum stacks).

Fluoride precipitation is another chemical reaction that occurs when infiltrating fluids and groundwater mix. The fluids from various EMF sources contain elevated fluoride concentrations compared to representative groundwater. For example, fluoride concentrations were over 1,500 mg/l in former Pond 8S fluids and over 6,000 mg/l in the former east overflow pond fluids, compared with 0.4 mg/l to 0.8 mg/l in representative groundwater. Equilibrium modeling indicates that the groundwater is nearly saturated with respect to fluorite at concentrations of about 0.75 mg/l and, therefore, conditions in the aquifer favor precipitation of the mineral fluorite (SII, 1994). Therefore, additional fluoride introduced into the aquifer cannot be held in solution by the groundwater, and it precipitates. The rapid decrease in fluoride concentrations can be seen at the former east overflow pond and former Pond 8S where fluoride concentrations decrease rapidly within a very short distance from the source.

Attenuation of fluoride can be confirmed by comparing the rate of its concentration decrease along the groundwater flowpath from the former ponds with total dissolved solids (TDS). TDS includes conservative solutes such as chloride and potassium for which any decrease in concentration with distance from a source is associated with advective mixing. The pronounced decrease in fluoride concentrations relative to TDS concentrations is an indication that fluoride precipitation occurred (Figure 5.3-3).

Under the pH conditions observed in groundwater at the gypsum stacks and other EMF sources, sulfate will precipitate from solution as gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) when concentrations exceed 2,500 mg/l (SII, 1994). With mean sulfate concentrations up to 2200 mg/l, the groundwater immediately beneath the gypsum stack is nearly saturated with respect to gypsum.

Eh also plays a role in the mobility of certain constituents in groundwater. For example, at former Pond 8S the two monitoring wells located immediately downgradient from the pond were the only wells in which reducing conditions were observed (Eh -150 mV). Oxidizing conditions prevailed in the aquifer upgradient and downgradient from the pond (+100 to +200 mV). When infiltrating fluid from this pond mixes with aquifer water, and Eh increases, there may be shifts in the valence state of certain chemicals that affect their solubility. These changes may reduce or increase mobility, depending on the chemical in question. Two reactions that were apparent downgradient from former Pond 8S are the transformation of ammonia to nitrate and total phosphorus to orthophosphate. Both these reactions reflect a shift from reducing to oxidizing conditions as the infiltrating fluid mixes and equilibrates with the groundwater.

The mobility of arsenic in groundwater was evaluated using thermodynamic principles. An analysis developed with the MINTEQA Model (Sciences International, 1994) was previously submitted to EPA. The analysis found "...that arsenic is relatively mobile in the EMF-site aquifer system because its migration potential is not effectively limited by any established chemical processes." (Reference Sciences International, 1994, page 4.12.) The analysis also found that arsenic concentrations were correlated with sulfate concentrations in groundwater, and that

arsenic concentrations showed "...a substantial decrease with increasing pH, suggesting that arsenic may be partially incorporated into a solid phase as the groundwater is neutralized." (Reference Sciences, 1994.) Sciences suggested that conditions favorable to the formation of tricalcium phosphate $[Ca_3(PO_4)_2]$ and/or strengite $[FePO_4 \cdot 2H_2O]$ exist in the EMF groundwater system, and that these might account for the observed decrease in phosphate concentration with increasing pH. "The substitution of a small mole fraction of AsO_4^{3-} for PO_4^{3-} in either of these two phosphate minerals may be the cause of the observed decrease in arsenic concentration with increasing pH." (Reference Sciences, 1994).

This analysis is supported by certain empirical data. In Wells 333, 307, and 308 (proximal to the gypstack), sulfate was detected at mean concentrations of 2,200; 1,912; and 1,962 mg/l, respectively. Mean arsenic concentrations in these wells were 0.34, 0.51, and 0.47 mg/l, respectively. In downgradient Wells 304 and 142, sulfate was detected at mean concentrations of 800 and 1,100 mg/l, respectively. The mean arsenic concentrations in these wells were 0.011 mg/l and 0.004 mg/l, respectively. The mean sulfate concentrations in Wells 304 and 142 were approximately one-half of the mean sulfate levels in Wells 333, 307, and 308. If arsenic were a non-attenuated solute, the mean arsenic concentrations in Wells 304 and 142 would be expected to be approximately one-half of the mean concentrations in Wells 333, 307, and 308 (i.e., 0.17 to 0.25 mg/l). Rather, the mean arsenic concentrations in Wells 304 and 142 were 1 to 4 percent of the mean arsenic concentrations in Wells 333, 307, and 308.

These wells are in the lower permeability region of the Bannock Range. This suggests that arsenic attenuation may be rate-limited in this area, perhaps through the mechanism suggested by Sciences' analysis.

5.3.2.2 Advective Mixing

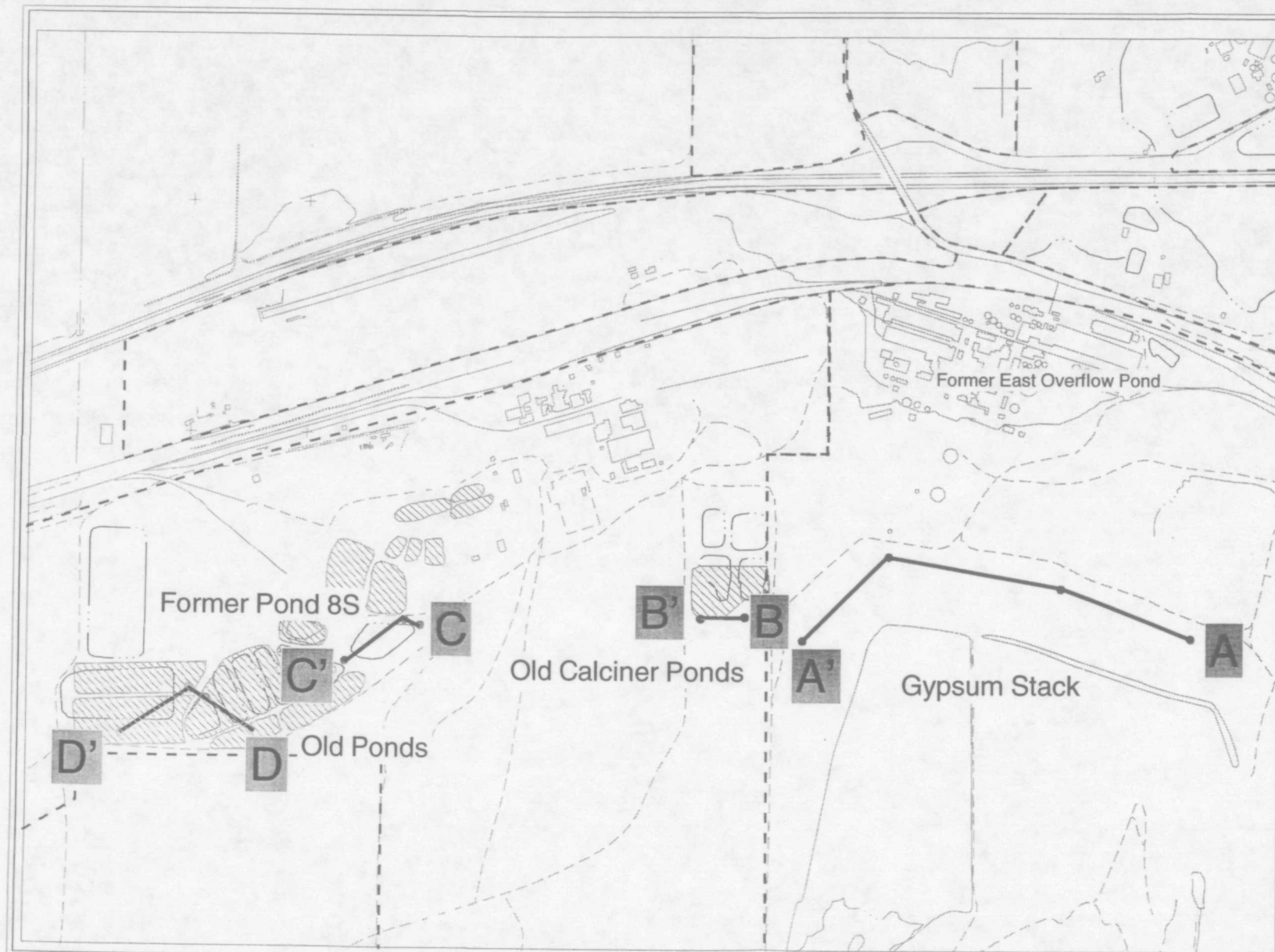
The most significant factor affecting the fate and transport of EMF-related constituents in groundwater is advective mixing. Advective mixing occurs when water with high solute concentrations mixes with water having low concentrations. The mixing results in a decrease of

the high solute concentrations. This can be illustrated by tracking the concentration of a conservative solute along a groundwater flowpath. Decreasing concentration with distance from a source is a good indication that advective mixing occurs. The rate of this decrease, or slope of the line defined by distance (x) versus concentration (y), provides a qualitative assessment of the advective mixing factor.

Sulfate is a conservative solute introduced into the aquifer from most of the EMF source areas. Its introduction into groundwater increased sulfate concentrations in wells immediately downgradient of source areas by 4 to 30 times the representative concentrations. However, these elevated concentrations often decreased to levels one to two times the representative concentrations within several hundred feet downgradient from these sources. As an example, peak sulfate concentrations in groundwater at the former east overflow pond are approximately 2,500 mg/l. Sulfate concentrations decreased to 280 mg/l, or by a factor of 9, within 300 feet downgradient from the pond (Figure 5.3-4).

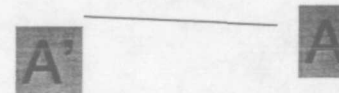
In summary, mixing of small volumes of affected groundwater with large volumes of unaffected groundwater within the EMF aquifer system substantially reduces the concentrations of all constituents, including conservative, non-attenuated solutes such as sulfate, along the groundwater flowpaths. This is further illustrated via the groundwater solute flux analysis presented in Section 5.4.

Surface and Subsurface Characterizations
Figures for Section 5.3

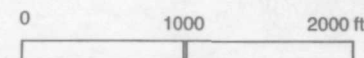


EXPLANATION:

Profile line



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EASTERN MICHAUD FLATS
POCATELLO, IDAHO

Soil Chemistry Profiles



Job Number

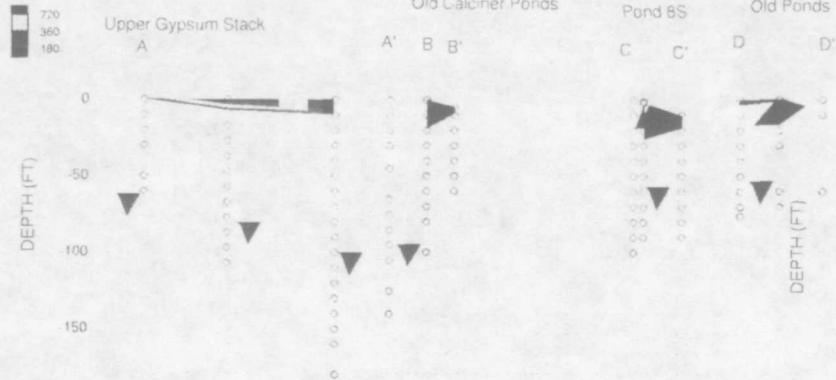
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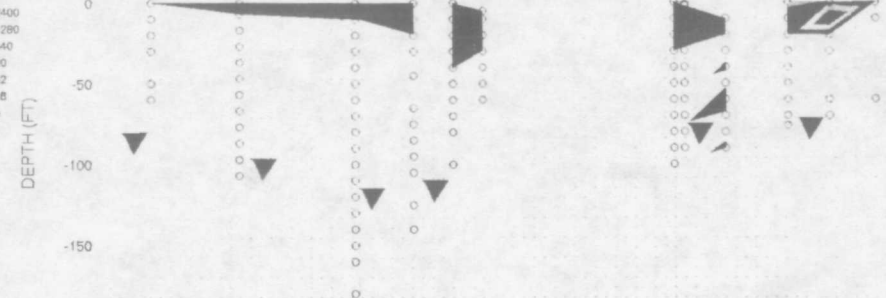
FIGURE 5.3-1a

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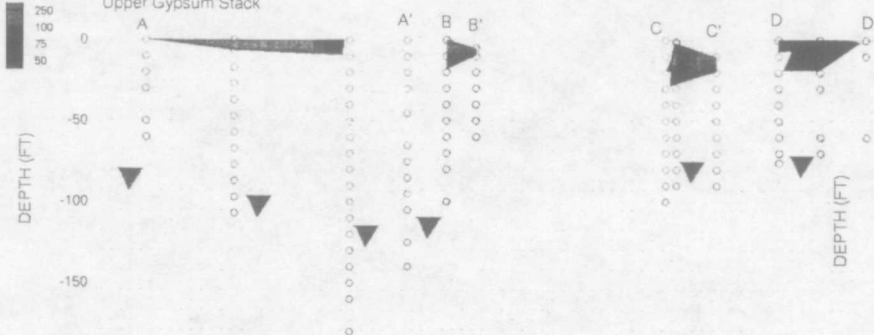
VANADIUM



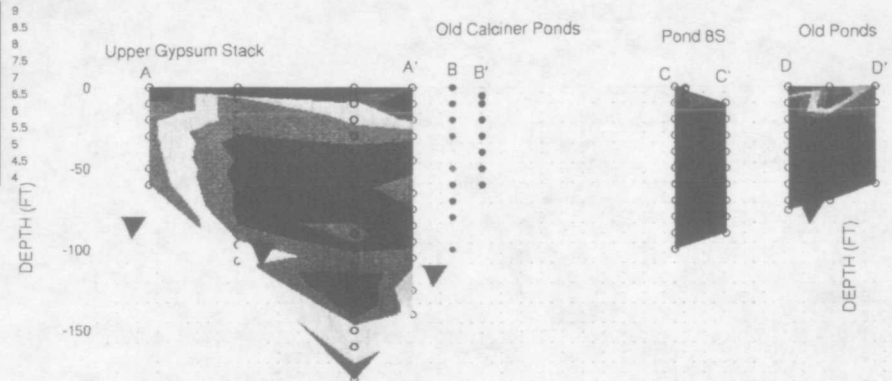
CADMIUM



CHROMIUM



pH



Note: pH measurements not collected for all soil samples at Old Calciner Ponds

▼ Approximate Depth of Groundwater

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Soil Chemistry Profiles



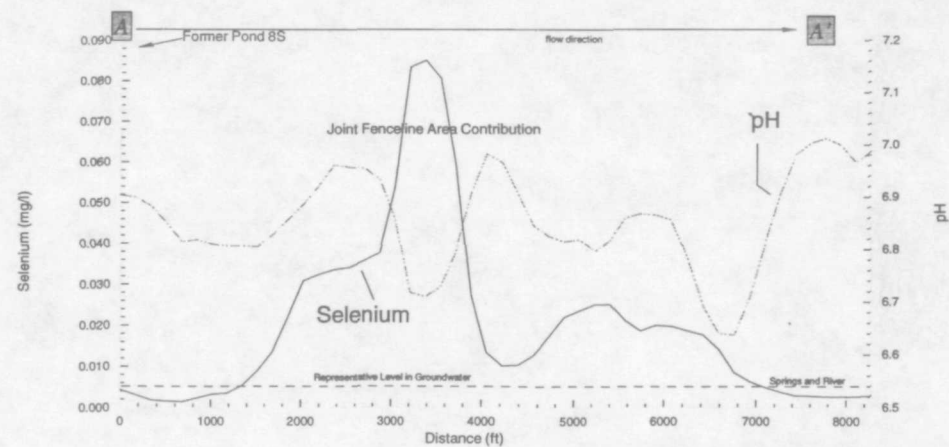
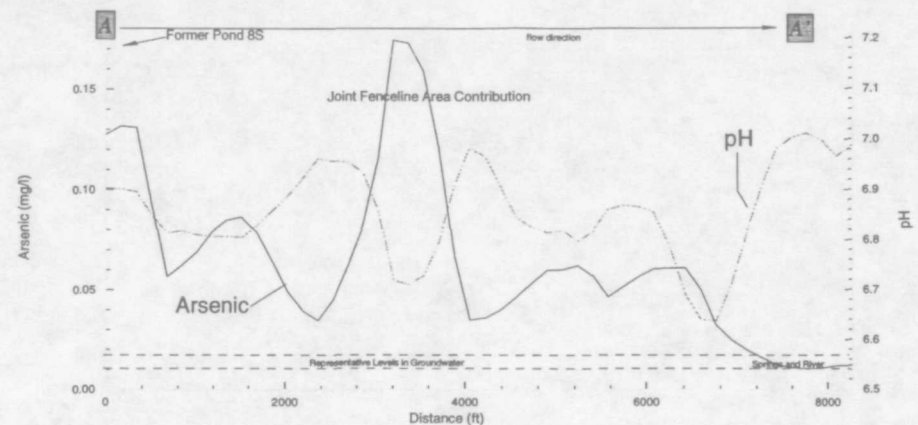
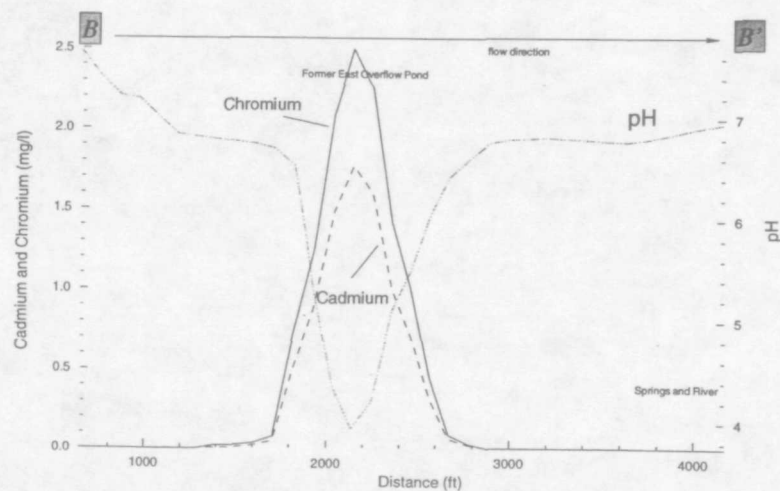
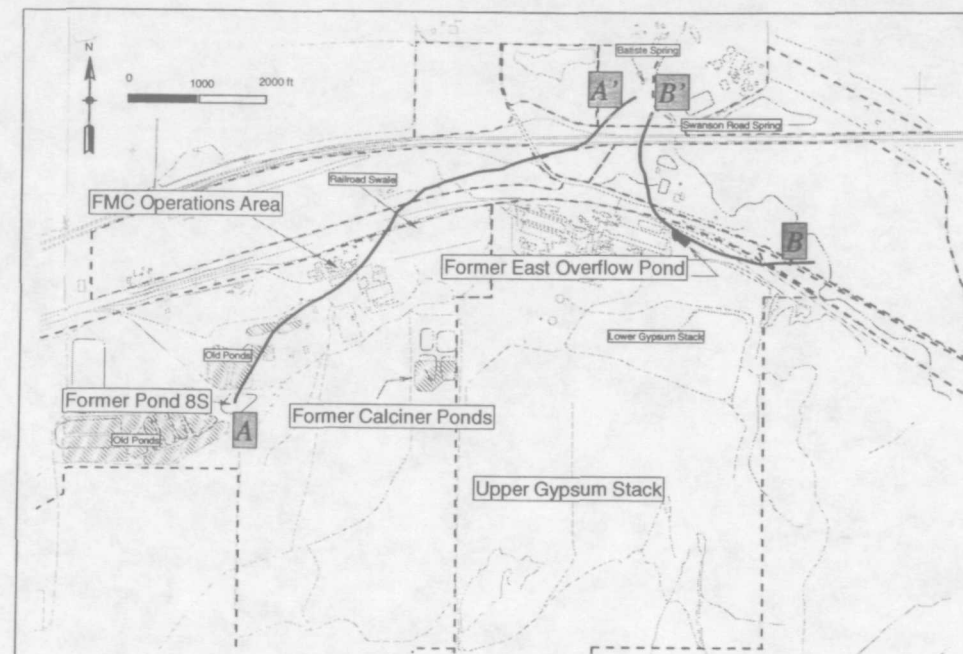
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FIGURE 5.3-1b

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Note: Concentration profiles from shallow groundwater concentration contour plots presented in Section 4.4. Concentration profiles were generated along particle tracks from the groundwater flow model (Appendix K).

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pH/Solute Relationships Along
Shallow Aquifer Particle Tracks



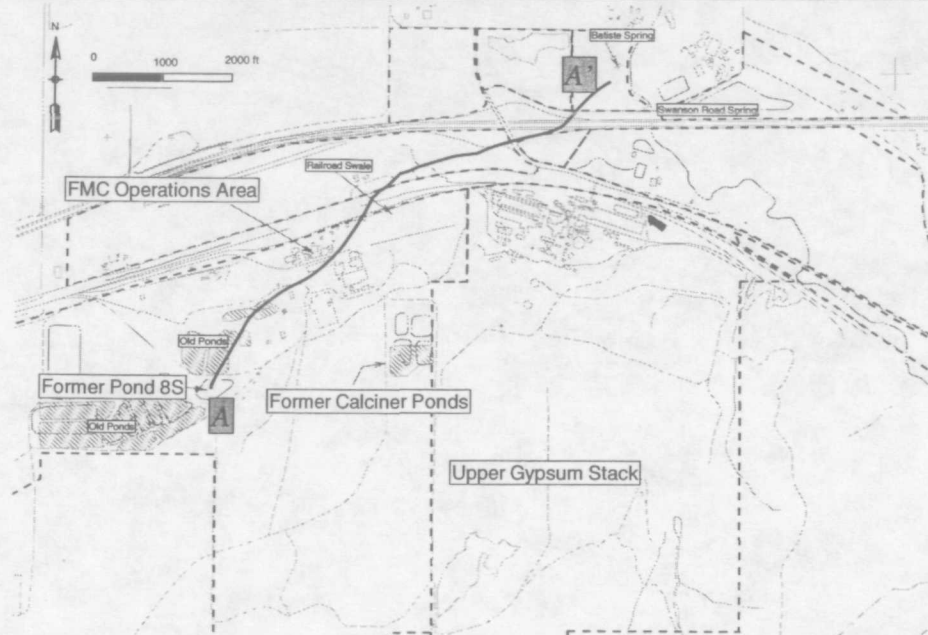
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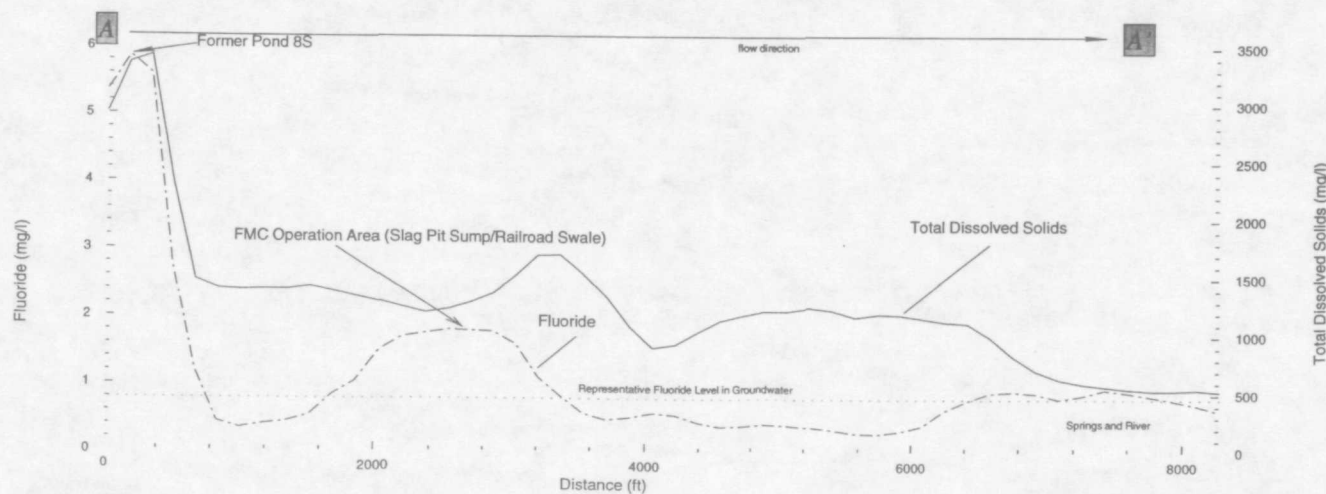
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FIGURE 5.3-2

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Note: Concentration profiles from shallow groundwater concentration contour plots presented in Section 4.4. Concentration profiles were generated along particle tracks from the groundwater flow model (Appendix K).



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Attenuation of Fluoride Along
Shallow Aquifer Particle Tracks



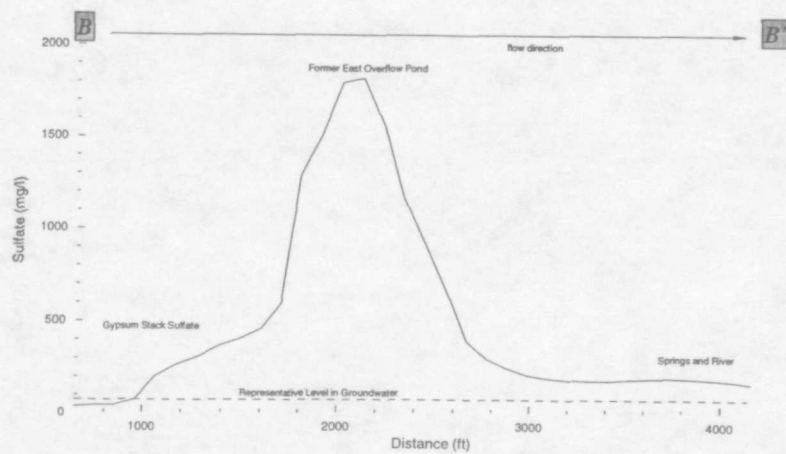
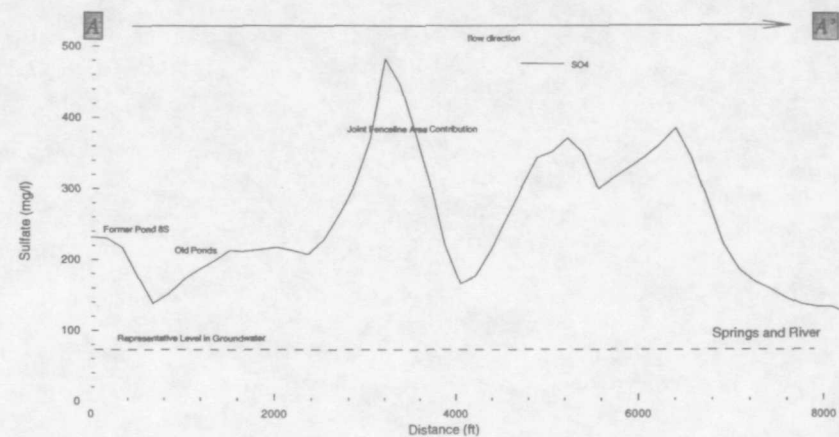
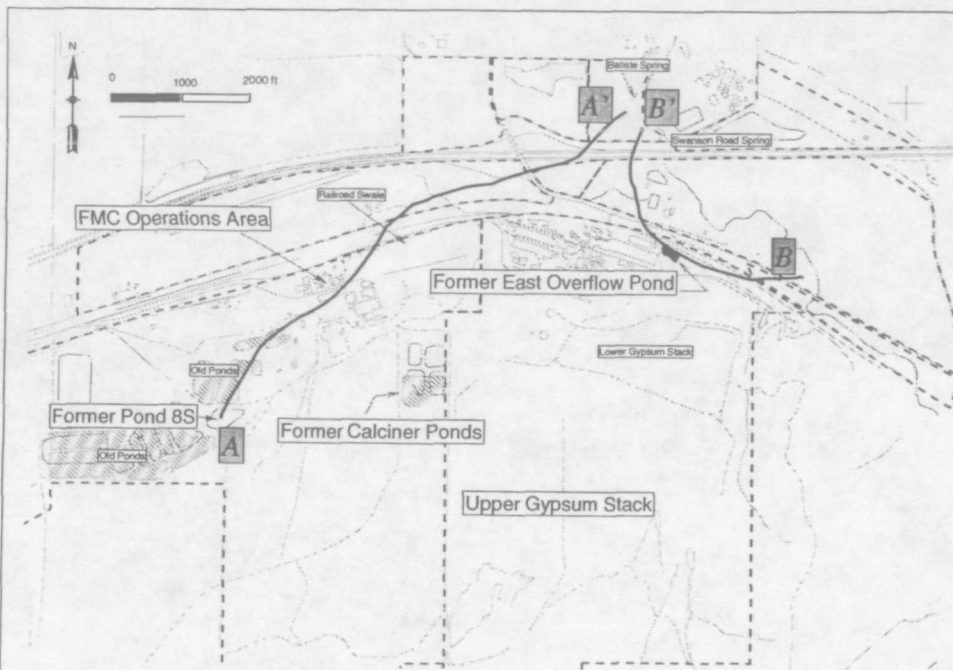
Job Number

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FIGURE 5.3-3

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Note: Concentration profiles from shallow groundwater concentration contour plots presented in Section 4.4. Concentration profiles were generated along particle tracks from the groundwater flow model (Appendix K).

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POCATELLO, IDAHO

Advective Mixing of Sulfate Along
Shallow Aquifer Particle Tracks



Job Number

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Drawing No.

FIGURE 5.3-4

Rev.

5.4 GROUNDWATER CONSTITUENT TRANSPORT AND DISCHARGE

This section describes the fate and transport of EMF-related constituents present in groundwater from the facilities to Batiste and Swanson Road Springs and the Portneuf River. Transport is described in terms of two types of fluxes: constituent mass fluxes and groundwater fluxes.

Groundwater flux is the volume of water crossing a given cross-section per unit time.

Constituent flux is the mass of dissolved constituents transported by the water. EMF-related constituent fluxes and background constituent fluxes are compared. Flux calculations are described in Section 5.4.1. Constituent transport in groundwater is discussed in Section 5.4.2. The influence of groundwater discharge on Portneuf River water quality is discussed in Section 5.4.3.

The analysis addresses the transport of arsenic, nitrate, selenium, sulfate, and total phosphorus. These constituents were identified by EPA and its risk assessment contractor as constituents of potential concern.

As indicated in Section 3.3, groundwater flowpaths from potential source areas at the EMF facilities converge in the area between and including Swanson Road Spring and Batiste Spring north of I-86. The fluxes of EMF-related constituents transported along these flow paths were calculated for two aquifer cross-sections. The trace of the first cross-section, which is generally parallel to Highway 30 north of the industrial production areas, is the position closest to the EMF facilities across which constituent fluxes from all potential source areas can be calculated. The second aquifer cross-section is downgradient from the first and roughly parallels the Portneuf River (Figure 5.4-1).

Fluxes were also calculated for upstream (losing) and downstream (gaining) reaches of the river and were compared with the EMF-related fluxes along the zone of groundwater discharge.

"Background and other non-EMF source" fluxes downstream of the EMF facilities were calculated by difference (i.e., total downstream fluxes minus EMF-related fluxes and upstream fluxes).

The flux analysis shows that the mass of EMF-related constituents discharged to surface water is small compared with background contributions. In addition, the analysis shows that EMF-related fluxes have relatively little influence on surface water chemistry. This finding is confirmed by the empirical data presented in Section 4.5.

Due to closures of unlined ponds since the 1992-1993 RI period of investigation, groundwater quality will improve over time. Thus, future EMF-related constituent fluxes are likely to be lower than those presented in this section.

5.4.1 FLUX CALCULATIONS

The mass flux of a solute across an area is the product of the solute concentration (mass / volume) and the water discharge through the area (volume / time). The units for the product are in mass / time. EMF solute fluxes were calculated in terms of kilograms (mass) per minute (time) or kg/min. Solute fluxes were calculated using defined aquifer and river channel cross sections, groundwater and surface water solute concentrations summarized in Sections 4.4 and 4.5, and data from the groundwater flow model (Appendix K).

5.4.1.1 Aquifer Cross-Sections

Two shallow aquifer cross-section lines were selected from the flow model grid (Figure 5.4-1). Cross-section line #1 starts between Wells 516 and TW-9S, where unaffected Michaud Flats groundwater and EMF-related groundwater converge and mix. The section line proceeds south to a point just north of Well 146 and then turns east, extending to a point just beyond the Frontier Well. The latter point was selected based on the flowpaths delineated from the flow modeling study and flowpaths plotted manually from groundwater elevation data. This section encompasses all shallow impacted groundwater from the EMF site as defined by estimated flowpaths and chemistry data from Wells 516, TW-9S, 146, 110, 312, 331, 320, 327, and the Frontier well.

Cross section line #2 is located along the west bank of the Portneuf River, where EMF-related groundwater discharges to springs and the river. The northernmost point of this section corresponds to Well 525, located approximately 150 feet north of Batiste Spring and it terminates just north of cross-section #1, near the Frontier well. This cross-section encompasses those regions of the shallow aquifer that are known to be transporting EMF-related constituents, as defined by modeled groundwater flowpaths and chemistry data from Wells 503, Frontier Well, Batiste and Swanson Road springs, as well as unaffected Wells 525 and 505.

5.4.1.2 Constituent Concentrations

Groundwater Concentrations

Analytical results from June 1992 through December 1993 groundwater sampling were used to calculate mean total solute concentrations at each well. Mean total solute concentrations for wells along cross sections #1 and #2 are presented in Table 5.4-1. Concentrations at most sampling locations did not change significantly during 1992-93, nor were they significantly different from concentrations detected for those wells in existence prior to this period. The absence of trends is supported by the analyses performed by FMC as part of its RCRA monitoring program (Bechtel, 1993g, 1994n, 1995). Therefore, these concentrations are believed to be representative of long-term, average concentrations.

Surface Water Concentrations

Analytical results for samples collected in four sampling events from July 1992 through April 1993 were used to estimate mean total concentrations. Upstream mean concentrations were calculated by averaging the results of samples collected at upstream (losing reach) stations (SW18-SW25) for the four sampling events. Samples collected at SW16, located at the I-86 bridge, and SW17, near the FMC IWW ditch outfall, were not used to characterize background fluxes in the Portneuf River because they are within the area of influence of the EMF facilities. Downstream mean concentrations were calculated by averaging all results from the downstream

sampling stations SW01, SW03, SW7E, SW08, SW10, SW12, and SW12E. The resulting means are presented in Table 5.4-2.

5.4.1.3 Groundwater Discharge

As described in Section 3.3 and Appendix K, average hydrogeologic conditions prevail at the EMF site. Groundwater flow direction, gradients, contour patterns, water levels, and flow rates have not changed appreciably on a seasonal, annual, or longer time scale. Therefore, the average groundwater flow patterns and flow rates described in Section 3.3 were used to estimate solute fluxes through the aquifer system during 1992-1993.

Groundwater fluxes across the aquifer cross-sections were calculated for the two uppermost model layers investigated in the groundwater flow model. (See Appendix K for description of flow model.) Model layers 1 and 2 incorporate the hydraulic parameters of the shallow aquifer, where most of the groundwater impacts have been observed. Consequently, EMF-related solutes are transported primarily through model layers 1 and 2. Groundwater in the deeper aquifer flows upward into the shallow aquifer as a result of upward vertical gradients. For example, certain constituent levels in groundwater flowing beneath the former east overflow pond in the deeper aquifer exceeded representative levels; however, the particle tracking results indicate that this deeper groundwater migrates vertically into the shallow aquifer upgradient from cross-section #1. The constituent mass transported along this flowpath is included in the flux calculated for each cross-section.

Model layers 3 and 4 include the widespread American Falls Lake Beds (regional aquitard) and the deeper aquifer zone (Appendix K). Horizontal groundwater fluxes across these layers were not estimated since there is little, if any, off-site transport of EMF-related solutes through these zones. This is supported by three lines of evidence:

- (1) Some deeper groundwater is captured by the Simplot production wells.

- (2) The solute concentrations in the deeper aquifer are typically much less than the concentrations detected in shallow monitoring wells, indicating there is no, or much less EMF-related impact to the deeper aquifer.
- (3) The upward gradients direct constituents present in the deeper groundwater in the south, near the gypsum stack, upward into the upper gravel zone. This flowpath is supported by the observed upward gradients in shallow/deep well pairs. The flow model particle tracks illustrate the effects of these vertical gradients, where the particle beginning in layer 4 flows upward into layer 2 near the transition from Bannock Range lithologies to Portneuf River lithology (Figure 3.3-16).

Cell-by-cell groundwater fluxes used in the calculations were taken from modeling scenario #1; this scenario assumes current operation of the gypsum stack, with a steady-state recharge rate of 500 gallons per minute, and operation of Simplot production wells SWP-5 and SWP-7 with a combined withdrawal rate of 3,500 gpm from the deeper aquifer.

Groundwater fluxes were obtained from the flow model output by extracting the flow across each cell face along the aquifer cross section. Because there are six faces per cell (top, bottom, front, rear, left, and right), only the flow component normal to the cross section was used. The dimensions of the cells physically describe the area of the cross section in square feet (top elevation, bottom elevation, width, and length).

Cross section #1 has a groundwater discharge of 7,600 l/min (4.5 cfs). At cross section #2, approximately 35,000 l/min (21 cfs) of groundwater discharges. The larger flow through cross-section #2 is attributable to a much higher transmissivity in the shallow aquifer. In addition, the upward flow component from the deeper aquifer to the shallow aquifer, coupled with the pinching out of the intervening aquitard, combine to increase the groundwater flow through cross-section #2.

5.4.1.4 Solute Fluxes

Groundwater

The solute mass transported across each model cell was estimated by multiplying the aquifer discharge through the cell by the mean solute concentration assigned to each cell. These were summed over the entire aquifer cross section to yield a total mass flux across this cross-section. Since groundwater fluxes are modeled on a cell-by-cell basis in the groundwater flow model, the mean solute concentrations were also assigned on a cell-by-cell basis. Mean concentrations were estimated by linear interpolation between the wells and springs located along the aquifer cross-section. For example, at Well 146, the mean sulfate concentration was 132 mg/l, and at Well TW-9S the mean sulfate concentration was 178 mg/l. Cells between these two points were assigned intermediate mean sulfate concentrations, with lower concentrations assigned to cells near Well 146 and higher concentrations assigned to cells nearer Well TW-9S. Appendix K (Section 3) provides further information on the selection of modeling modules.

The EMF-related solute mass was estimated by subtracting the background solute flux from the total flux. The background solute flux was calculated using the representative groundwater concentrations for the area in question (Bannock Range or Portneuf River area). Bannock Range groundwater flows through cross-section #1. Representative levels for each constituent in Bannock Range groundwater were multiplied by the groundwater flow across each cross-section to determine the background flux for cross-section #1. At cross-section #2, Portneuf River groundwater, Michaud Flats groundwater, and Bannock Range groundwater mix and discharge to the river. Representative constituent concentrations for Portneuf River groundwater were used to quantify the background solute contributions across cross section #2. This is because Portneuf River water is observed discharging at Swanson Road Springs. The representative concentrations for groundwater in the three hydrogeochemical regimes are summarized in Table 5.4-3.

Portneuf River

The solute mass transported across the cross-sectional area of the river at the Carson and Tyhee gauges was estimated by multiplying the average river discharge by the mean solute concentration (Tables 5.4-2 and 5.4-4). Upstream mean concentrations were multiplied by the average river discharge at the Carson Street gauge (310 cfs). Downstream mean concentrations were multiplied by the average discharge at the Tyhee gauge (516 cfs). River discharges are described in more detail in Section 3.2 and Appendix K.

5.4.2 DISCUSSION – CONSTITUENT TRANSPORT IN GROUNDWATER

This section describes the influence of EMF-related sulfate, total phosphorus, nitrate, arsenic, and selenium on groundwater quality by comparing EMF-related fluxes to background fluxes at cross sections #1 and #2 (Table 5.4-5).

5.4.2.1 Sulfate

The sulfate mass flux at cross-section #1 was approximately 3.0 kg/min. With a background sulfate concentration of 43 mg/l, the flux of naturally occurring sulfate was approximately 0.3 kg/min. The net EMF-derived contribution of sulfate was thus 2.7 kg/min (Table 5.4-5; Figure 5.4-2).

Sulfate mass flux at cross-section #2 was 5.1 kg/min. Background sulfate accounted for 2.4 kg/min using a background concentration of 70 mg/l (Table 5.4-5). EMF-related sulfate was 2.7 kg/min (Table 5.4-5; Figure 5.4-2). As noted above, the calculated mass of EMF-related sulfate across cross-section #1 was also 2.7 kg/min. The mass of EMF-related sulfate transported off-site is accounted for at both aquifer cross sections.

5.4.2.2 Arsenic

The flux of EMF-related arsenic across aquifer cross-section #1 was 0.00029 kg/min, whereas the estimated flux of EMF-related arsenic transported across cross-section #2 was

0.00020 kg/min (Table 5.4-5; Figure 5.4-2). Given the low constituent concentrations and, hence, low total mass of arsenic in the groundwater system, the estimated fluxes appear to be within a reasonable range of agreement. This illustrates that, even for constituents introduced into the aquifer system at much lower concentrations than sulfate, the mass transport calculations are in reasonable agreement.

5.4.2.3 Selenium

EMF-related selenium transported across cross-section #1 was calculated to be 0.00015 kg/min. Background selenium contributed an additional 0.00004 kg/min at this cross-section (Table 5.4-5; Figure 5.4-2). At cross-section #2, the EMF-related flux of selenium was 0.000032 kg/min. The flux of EMF-related selenium into the Portneuf River can be distinguished from the background flux only by using the most conservative (i.e., lowest) estimated background value for selenium (Table 5.4-5). The representative level of selenium in Portneuf River groundwater is 0.006 mg/l. If this value is used to estimate the EMF-derived selenium contribution to the Portneuf River, the net result is a negative contribution. If a background selenium concentration of 0.004 mg/l is used, the EMF-derived contribution is 0.000032 kg/min. The estimated selenium fluxes are not in close agreement, due to the low selenium concentrations and, hence, low total mass of selenium in the groundwater system.

5.4.2.4 Total Phosphorus

The calculated mass of total phosphorus did not change appreciably from cross section #1 to cross-section #2 (Table 5.4-5; Figure 5.4-2). Mass transport calculations indicated that approximately 0.065 kg/min of EMF-related phosphorus was transported past cross-section #1 with an 8 percent decrease at cross-section #2.

5.4.2.5 Nitrate

The EMF-related nitrate mass transported through cross-section #1 was 0.055 kg/min during the 1992-1993 period of investigation. This was two times the mass of nitrate accounted for at cross-section #2 – 0.028 kg/min (Table 5.4-5; Figure 5.4-2). The overall reduction of nitrate mass transported through the aquifer system may be explained, in part, by biological nitrogen fixation.

5.4.3 DISCUSSION – INFLUENCE OF GROUNDWATER DISCHARGE ON SURFACE WATER

This section describes the potential influence of the discharge of EMF-related sulfate, total phosphorus, and nitrate on Portneuf River water quality during the 1992-1993 period of investigation. This is done through a comparison of EMF-related groundwater solute fluxes with background groundwater fluxes. The constituent concentrations in surface water that are estimated to result from these EMF-related discharges are compared with concentrations detected in surface water during the RI. This analysis was not performed for arsenic and selenium, because, as indicated in Section 4.5, no measurable EMF arsenic or selenium effects were observed in surface water.

The mass loading of EMF-related constituents to the Portneuf River during 1992-1993 is the mass flux of EMF-related constituents passing through aquifer cross-section # 2. The additional contribution of constituents from the FMC outfall detected in September 1992 (Section 4.2.3) was due to an unintentional, short-term release confirmed by subsequent sampling. Thus, the flux potentially attributable to this release has not been included in the analysis.

The influence of EMF-related discharges was evaluated under two conditions: average conditions and low river-flow conditions, the latter being considered worst-case. The worst-case analysis used the lowest river flow rate recorded during the RI. This low flow condition occurred in July 1992, during an RI surface water sampling event where upstream river discharge was 18 to 20 cfs, compared to the average of 310 cfs.

5.4.3.1 Sulfate

Average sulfate fluxes in the river increased markedly between the upstream river reach and the downstream reach (Table 5.4-4). Upstream from the EMF site, approximately 22.8 kg/min of sulfate are transported along the river. EMF-related sulfate flux is 2.7 kg/min, and the Pocatello STP discharge is estimated at 2.1 kg/min based on an average sulfate concentration of 123 mg/l (Wood and Low, 1986) and discharge of 11 cfs (City of Pocatello, 1989). The downstream river transports approximately 53.3 kg/min of sulfate. The increase of 30.5 kg/min in sulfate flux is mostly a function of the background level of sulfate throughout the aquifer, which discharges almost 200 cfs of water to the river, rather than a function of sulfate from the EMF facilities or the Pocatello STP.

Under average conditions, the estimated river sulfate concentrations are 43 mg/l upstream from the site and 60 mg/l downstream from the site (Table 5.4-6). The estimated EMF-related sulfate contribution to river concentrations is approximately 3.0 mg/l, whereas the estimated overall concentration increase is 17 mg/l. This indicates that while the EMF site might contribute about 3.0 mg/l in sulfate concentrations to the river water, 14 mg/l are contributed by background groundwater, the STP, and other discharges to the river.

The downstream average river concentration during the low flow conditions was 59 mg/l and the upstream concentration was 39 mg/l (Table 5.4-7). Upstream and downstream average concentrations are very similar to average concentrations. These results indicate that sulfate fluxes from the EMF site groundwater and the STP do not have a greater measurable impact on the river during low flow conditions.

5.4.3.2 Total Phosphorus

Total phosphorus fluxes in the Portneuf River increase by a factor of 10 from upstream to downstream. Total phosphorus flux in the upstream river reach was estimated to be 0.053 kg/min at average concentrations of 0.10 mg/l. Downstream from the site, concentrations

average 0.59 mg/l and the flux is 0.52 kg/min. EMF-related total phosphorus flux was 0.06 kg/min, or 11% of the flux as measured in the downstream river reach. Background sources contributed a greater flux (0.417 kg/min). The estimated increase in total phosphorus concentrations in river water from EMF-related discharge was 0.069 mg/l. Other contributors of total phosphorus to the river include the fish farms, the STP, and groundwater unaffected by the EMF facilities.

The average downstream concentration is 0.59 mg/l, and during worst-case conditions the average downstream concentration was 0.50 mg/l. EMF-related effects were not measurable during the low flow conditions. It appears that neither the STP, fish farms, nor the EMF facilities measurably impacted surface water quality during low river flow conditions. Rather, the discharge of background groundwater, with representative concentrations on the order of 0.15 to 0.33 mg/l, appears to increase the total phosphorus concentrations more than any other source.

5.4.3.3 Nitrate

Nitrate fluxes in the Portneuf River also show a significant increase between upstream and downstream reaches (Table 5.4-4). Upstream from the EMF facilities, the river transports 0.32 kg/min of nitrate at average concentrations 0.60 mg/l. Downstream from the facilities, approximately 1.8 kg/min at an average concentration of 2.07 mg/l are transported by the river. EMF-related nitrate discharge to the river was estimated to be 0.028 kg/min, or less than 2% of the total downstream nitrate flux in the river.

The EMF facilities may contribute about 0.0325 mg/l in nitrate concentration to the river water. The primary contributor of increased nitrate in the river is non-EMF-related groundwater discharge. Approximately 200 cfs of background groundwater discharges to the river, transporting with it about 3 to 5 mg/l nitrate. The STP also discharges measurable amounts of ammonia, which may be converted to nitrate in the presence of dissolved oxygen.

The worst-case, or low-flow, impacts do not appear to be different than the average conditions (Table 5.4-7). Downstream nitrate concentrations were not different during the low-flow conditions when compared to average conditions. In July 1992, during low flow conditions, there was an increase of 1.98 mg/l from upstream to downstream. Average conditions reflect an increase of 1.47 mg/l, similar to the worst-case condition.

5.4.4 UNCERTAINTY ASSESSMENT

The solute mass flux from EMF-related sources was estimated by defining the region of the aquifer that transported EMF-derived constituents toward the discharge area along the Portneuf River during the period of investigation, extracting the groundwater flux through this portion of the shallow aquifer, and multiplying the groundwater flux by the solute concentrations. Solute concentrations were defined by average concentrations in shallow wells located along two aquifer cross-sections. The solute mass contributed by background solutes was subtracted from the total solute flux and the remaining flux was assumed to be from all EMF sources affecting groundwater.

Solute flux was calculated at two aquifer cross sections, rather than at one, to minimize the uncertainty in estimating the mass transport of EMF-derived constituents to the Portneuf River. Cross section #1 is close to and downgradient of all potential source areas within the EMF facilities; cross section #2 is further downgradient and near the Portneuf River. The calculated EMF-derived solute mass at each cross section (see Table 5.4-5) showed good agreement for sulfate and arsenic (0% difference for sulfate and a 33% decrease from cross section #1 to cross section #2 for arsenic). Greater differences were calculated for nitrate, total phosphorus, and selenium. However, the EMF-related portion of these solute fluxes was consistently lower at cross section #2.

Comparison of the sulfate fluxes at these cross sections is a measure of the accuracy of the identification of EMF-derived constituent fluxes. Sulfate is a conservative ion. If its flux at cross section #2 were greater than the flux at cross section #1, the calculated flux of this and other EMF-derived constituents may have been underestimated. This was not the case.

The calculated EMF-derived fluxes of arsenic, nitrate, total phosphorus, and selenium at cross section #2 were less than their flux at cross-section #1. This indicates that the flux of these EMF-derived constituents at cross section #1 was not underestimated. These constituents may have been attenuated in the area between these cross-sectional lines. Alternatively, the background solute flux at cross-section #2, or the flux of these EMF-derived solutes at cross-section #1, may have been overestimated.

The solute mass flux estimates used in these comparisons derives from two terms: groundwater flux and solute concentrations. Uncertainties associated with the groundwater flux are derived from the numerical modeling input parameters and are discussed in Section 5.4.4.1.

Uncertainties associated with the solute concentrations used to calculate solute fluxes are discussed in Section 5.4.4.2. A summary is provided in Section 5.4.4.3.

5.4.4.1 Sources of Uncertainty in Groundwater Flux Estimates

Terms contributing to the flux at cross-section #1 are the underflow through the southern model boundary, background recharge, gypsum stack recharge, Pond 8S recharge and a portion of the underflow from the southeastern model boundary. These terms contribute 4.5 cfs of discharge at cross-section #1. The southern model boundary and the gypsum stack contribute 4.2 cfs combined. The remaining three components make up 0.3 cfs or less than 10% of the total discharge.

At cross-section #2, the flux was 21 cfs. Terms contributing to this flux are those listed for cross-section #1 plus significant contributions of underflow from the western and eastern model boundaries (this boundary underflow ultimately discharges at cross-section #2 due to the upward movement of groundwater where the AFLB aquitard is not present). Some river water is lost to the shallow aquifer through a portion of the model domain, and this water flows across cross-section #2. Since the solute fluxes at this cross section were used as a check against those

calculated at cross-section #1, the groundwater fluxes and model uncertainties will not be discussed for cross-section #2.

Southern Boundary Underflow. The flux of groundwater flowing beneath the EMF source areas is a function of the constant heads along the southern boundary and the hydraulic conductivity zones adjacent to this boundary. Groundwater flux entering the model domain is controlled by the hydraulic conductivity and boundary heads that were assigned to boundary cells.

The boundary heads were set using heads measured at 4 wells, whereas hydraulic conductivity measurements along the southern model boundary consisted of one slug test and one pumping test. Consequently, there is a larger uncertainty associated with the hydraulic conductivity and the modeled fluxes will be more sensitive to the hydraulic conductivity assigned to the cells along the southern boundary. The model was not sensitive to the hydraulic conductivity in the Bannock Range within a factor of 5. In other words, the hydraulic conductivity could be varied about the measured values by +/-5 times and the model would still meet most calibration targets. This insensitivity was accommodated in the flow model by setting the southern boundary flux term as a calibration target. This target ultimately restricted hydraulic conductivity along this boundary to a value that restricts the boundary flux to a reasonable level (i.e., that which can be accounted for by recharge from the Bannock Range).

The southern boundary underflow was estimated using analytical methods and a recharge term was assigned to this area. The small drainages along the southern model boundary within the Bannock Range have an area of about 4,000 acres, and this area was assumed to have an average annual precipitation of 1.5 times the average from the Pocatello Airport. This higher precipitation rate was used in the calculation because average precipitation is typically higher in the mountainous regions of the Western U.S. compared with valley floors. Groundwater flow divides were assumed to parallel surface water flow divides along the southern model boundary.

Recharge to groundwater was estimated to be 15% of the annual precipitation. This higher recharge rate was used because precipitation occurring as snow in the uplands may have a greater chance of infiltrating, and a greater amount of the precipitation occurs as snow in the Bannock Range. Using 4,000 acres and a recharge rate of 2.4 inches per year yielded an analytical solution for the southern boundary underflow of 0. 1.07 cfs.

One of the calibration targets for the numerical groundwater flow model was to be within a factor of three (i.e., between 0.35 to 4.2 cfs) of this analytically-derived southern model boundary flux. The modeled boundary flux was 3.1 cfs. This was within the calibration target range, but about 2 cfs greater than the calculated upper limit of 1.07 cfs ($3.1 \text{ [modeled]} - 1.07 \text{ [upper range of analytically-calculated]} = 2.04 \text{ cfs}$).

As discussed above, the groundwater flux through the aquifer cross-section #1 in the solute transport model was 4.5 cfs. The uncertainty associated with the southern boundary conditions could account for an overestimation of 2 cfs at this cross section. A larger groundwater flux through cross-section 1 will yield a higher estimate for the total solute flux along the cross section, and an overestimation of the EMF-derived solute flux. Therefore, an overestimation of the southern boundary groundwater flux would be conservative when estimating EMF-related solute mass transport.

Gypsum Stack Recharge. The gypsum stack recharge rate is a crude estimate; however, there is a known upper limit. Simplot pumps an average of 2300 gpm of slurry to the gypsum stack. There is a 30% solids content in the slurry, which means there are 1600 gpm of water available for recharge to the groundwater. It is likely that a significant portion of this water is evaporated prior to infiltrating to the subsurface or retained in the gypsum through hydration reactions.

For this study, an evaporation rate of 200–250 gpm was assumed and 500 gpm (1.1 cfs) was recharged to the groundwater from the gypsum stack. This estimate was deemed conservative for this modeling study, since steady state conditions were assumed and the true operation of the

gypsum stack may not actually approach steady state conditions. Actual gypsum stack operations may result in lower recharge because of: (1) the gypsum decant recovery system recovers water infiltrating through the stack from a series of lateral drains, and (2) applying slurry to different parts of the gypsum stack for a period of several weeks or months may not allow the system to approach equilibrium conditions. This practice of applying slurry may actually enhance evaporative losses, thereby reducing the amount of recharge. Further discussion of factors influencing recharge at the gypstack is presented in Appendix K (Groundwater Flow Modeling Report, section 2.1.2.2)

Background Recharge and Pond 8S Recharge. Pond 8S recharge was estimated from a detailed water budget study conducted at the pond in 1992 (Bechtel, 1993). This study determined that a water loss of 15.3 gpm via infiltration through the bottom of the pond was needed to maintain the water levels observed within the pond during the period of study. Background recharge is 10% of the average annual precipitation, or about 1.1 inch per year.

Groundwater flux across the model domain is not believed to be sensitive to either of these terms. Pond 8S contributed 0.033 cfs, and background recharge within the model domain, upgradient from cross-section #1, contributed about 0.20 cfs. An increase or decrease in these terms by 300% would increase or decrease the groundwater flux at cross-section #1 by no more than 10%.

5.4.4.2 Uncertainty in Solute Concentrations

Uncertainty in the solute fluxes is related to uncertainty in the mean solute concentrations and the interpolation of values between wells. Laboratory error, sampling error, temporal variations in concentrations, the assumption that concentrations vary in linear fashion between wells, and identification of representative concentrations all contribute to the uncertainty associated with the calculated solute fluxes.

The potential for laboratory and sampling error was minimized through adherence to the quality assurance procedures established in the field sampling plan and QAPP. The potential for sampling and lab uncertainty was within an acceptable range, as described in the assessment of the precision, accuracy, representativeness, completeness, and comparability (PARCC) of the analytical data (see Appendix J). This process minimized the potential for errors in the analytical data used in these calculations.

FMC has been performing time-series analyses at its RCRA wells since 1990. These analyses are reported in a series of annual groundwater monitoring assessment reports prepared under the RCRA interim status groundwater monitoring standards (FMC 1993, 1994, 1995, and 1996). The analysis of up to 15 wells indicates no significant trends in solute concentrations. Solute concentrations at the wells, springs, and river samples used in the solute mass transport study were relatively stable through the course of the RI (from 1992 through 1994). Since the solute fluxes were estimated at numerous points along each cross section (each model cell), the uncertainty in solute flux would be limited to the region between that well and adjacent wells along the cross-section. For example, there may exist preferential transport pathways between wells that were not monitored during this study, resulting in underestimation of solute transport. This type of uncertainty was minimized by using two cross-sections rather than one.

The estimation of representative concentrations could have introduced uncertainty into the estimates of EMF-derived solute mass fluxes. Representative groundwater concentrations of arsenic, nitrate, and sulfate varied significantly among the Michaud Flats, Portneuf River Valley, and Bannock Range groundwaters. The use of a lower representative concentration in computing solute flux would mean that a higher proportion of the total solute mass would be credited to EMF sources. The use of a higher representative value would mean that less of the total solute mass would be credited to EMF sources.

The representative value for selenium was very near the detection limit, reducing the likelihood of underestimation of these values. The representative value for arsenic in the Bannock Range groundwater (0.018 mg/l) was higher than the value in Portneuf River groundwater (0.010 mg/l).

The Portneuf River value was used in computing representative arsenic flux across cross-section #2. If the higher Bannock Range arsenic representative value had been used at this cross-section, the resulting EMF-related arsenic discharge to the springs and river would have been reduced by a factor of 1.8. Instead of 0.0002 kg/min of EMF-related arsenic discharging to the river, 0.00011 kg/min of EMF-related arsenic would have been calculated.

Total phosphorus representative concentrations were also higher in the Bannock Range groundwater, by a factor of 2, than the total phosphorus representative level in the Portneuf River groundwater. The representative level in the Portneuf River groundwater was used in calculating site-related total phosphorus flux crossing cross-section #2. If the higher representative level in the Bannock Range groundwater had been used, the mass flux of total phosphorus attributed to EMF sources would have been lower.

Representative sulfate concentrations were higher in the Portneuf River groundwater by a factor of 1.7 compared with Bannock Range groundwater. Nitrate was 2.5 times higher in the Portneuf River groundwater compared with Bannock Range groundwater. If the proportion of Bannock Range groundwater flowing across aquifer cross-section 2 is higher than previously estimated, the representative contributions for sulfate and nitrate were overestimated, and EMF-related contributions would be greater than those shown in Table 5.4-5.

5.4.4.3 Summary

The solute mass fluxes are most sensitive to uncertainty in the calculation of groundwater fluxes. These fluxes could have been greatly overestimated or slightly underestimated based on the analytically-derived range of Bannock Range recharge. Any overestimation in this flux would have a conservative effect in the calculation of site-related effects on groundwater and surface water quality. Based on this, the Companies have presented a scenario that is conservative, (i.e., more likely to overestimate than underestimate the EMF-related contributions).

The potential uncertainty associated with chemical concentrations at wells and springs is very low, due to the large amount and quality of data used in the assessment.

**Surface and Subsurface Characterizations
Tables for Section 5.4**

TABLE 5.4-1
MEAN CONSTITUENT CONCENTRATIONS

TABLE 5.4-1

CROSS SECTION #1									
Analyte ^(a) (mg/l)	Well 516	Well TW-9S	Well 146	Well 110	Well 312	Well 331	Well 320	Well 327	Frontier Well
<u>Major Ions</u>									
Sulfate	67.0	177.5	135	586	1,262	1,060	903	280	175.8
<u>Nutrients</u>									
Nitrate (NO ₃ as N)	1.97	12.64	5.37	2.67	4.62	30.90	66.0	2.43	1.81
Total Phosphorus	0.06	3.08	0.29	8.18	29.14	27.20	36.7	6.45	2.43
<u>Metals</u>									
Arsenic	0.0055	0.0352	0.0213	0.1197	0.2377	0.2325	0.0680	0.0292	0.0185
Selenium	0.0025	0.0022	0.0063	0.0686	0.1543	0.0681	0.0161	0.0039	0.0033

CROSS SECTION #2						
Analyte ^(a) (mg/l)	Well 525 ^(b)	Batiste Spring	Well 503	Swanson Road Spring	Well 505	Frontier Well
<u>Major Ions</u>						
Sulfate	50	115.22	238.4	105	65.6	175.8
<u>Nutrients</u>						
Nitrate (NO ₃ as N)	1.90	3.42	4.47	2.64	2.77	1.81
Total Phosphorus	NA	1.88	4.16	0.87	0.08	2.43
<u>Metals</u>						
Arsenic	0.0032	0.0150	0.0169	0.0069	0.0047	0.0185
Selenium	0.0035	0.0069	0.0039	0.0034	0.0028	0.0033

Notes: (a) Chemical concentrations shown on this table are mean values from data collected between June 1992 and December 1993.

(b) Well 525 installed in June 1994; mean concentrations are based on data collected in June and September 1994.

NA = Not analyzed.

TABLE 5.4-2
LONG-TERM SOLUTE CONCENTRATIONS IN PORTNEUF RIVER

CONSTITUENT	UPSTREAM MEAN CONCENTRATION ^(a) (mg/l)	DOWNSTREAM MEAN CONCENTRATION ^(b) (mg/l)
Nitrate	0.60	2.07
Sulfate	43	60
Total Phosphorus	0.10	0.59

^(a) Means calculated using July 1992 through April 1993 data for samples from locations SW18 through SW25.

^(b) Means calculated using July 1992 through April 1993 data for samples from locations SW01, SW03, SW7E, SW08, SW10, SW12, and SW12E.

**TABLE 5.4-3
REPRESENTATIVE GROUNDWATER CHEMISTRY**

ANALYTE (mg/l)	95% UPPER CONFIDENCE LEVEL FOR REPRESENTATIVE GROUNDWATER IN EACH HYDROGEOCHEMICAL REGIME ^(a)		
	Michaud	Bannock	Portneuf
<u>Major Ions</u>			
Sulfate	72.6	43.4	70.7
<u>Nutrients</u>			
Nitrate (NO ₃ as N)	5.52	1.60	4.00
Total Phosphorus	0.33	0.31	0.15
<u>Metals</u>			
Arsenic	0.0149	0.0180	0.0104
Selenium	0.0057	0.0055	0.0060

Notes: ^(a) The 95% UCLs are based on the total population of analytical results for representative groundwater.

TABLE 5.4-4
CALCULATED LONG-TERM SOLUTE FLUXES IN PORTNEUF RIVER

CONSTITUENT	UPSTREAM (kg/min)	EMF-RELATED (kg/min)	BACKGROUND, STP, AND OTHER SOURCES (kg/min)	DOWNSTREAM (kg/min)
Nitrate	0.32	0.028	1.45	1.80
Sulfate	22.8	2.7	27.8	53.3
Total Phosphorus	0.053	0.06	0.417	0.52

TABLE 5.4-5
GROUNDWATER SOLUTE FLUXES

CONSTITUENT	BACKGROUND (kg/min)	EMF-RELATED (kg/min)	TOTAL (kg/min)
<u>ARSENIC</u>			
Solute Flux at Cross Section #1	0.00014	0.00029	0.00043
Solute Flux at Cross Section #2	0.00035	0.00020	0.00055
<u>NITRATE</u>			
Solute Flux at Cross Section #1	0.012	0.055	0.067
Solute Flux at Cross Section #2	0.071	0.0280	0.099
<u>SULFATE</u>			
Solute Flux at Cross Section #1	0.3	2.7	3.0
Solute Flux at Cross Section #2	2.4	2.7	5.1
<u>SELENIUM</u>			
Solute Flux at Cross Section #1	0.000040	0.00015	0.00019
Solute Flux at Cross Section #2	0.00014	0.000032	0.00018
<u>TOTAL PHOSPHORUS</u>			
Solute Flux at Cross Section #1	0.0019	0.065	0.067
Solute Flux at Cross Section #2	0.011	0.060	0.071

Notes: Groundwater Flux at Cross Section #1: 7,600 l/min and 4.5 cfs
Groundwater Flux at Cross Section #2: 35,600 l/min and 21.0 cfs

TABLE 5.4-6
CALCULATED LONG-TERM SOLUTE CONCENTRATIONS IN PORTNEUF RIVER

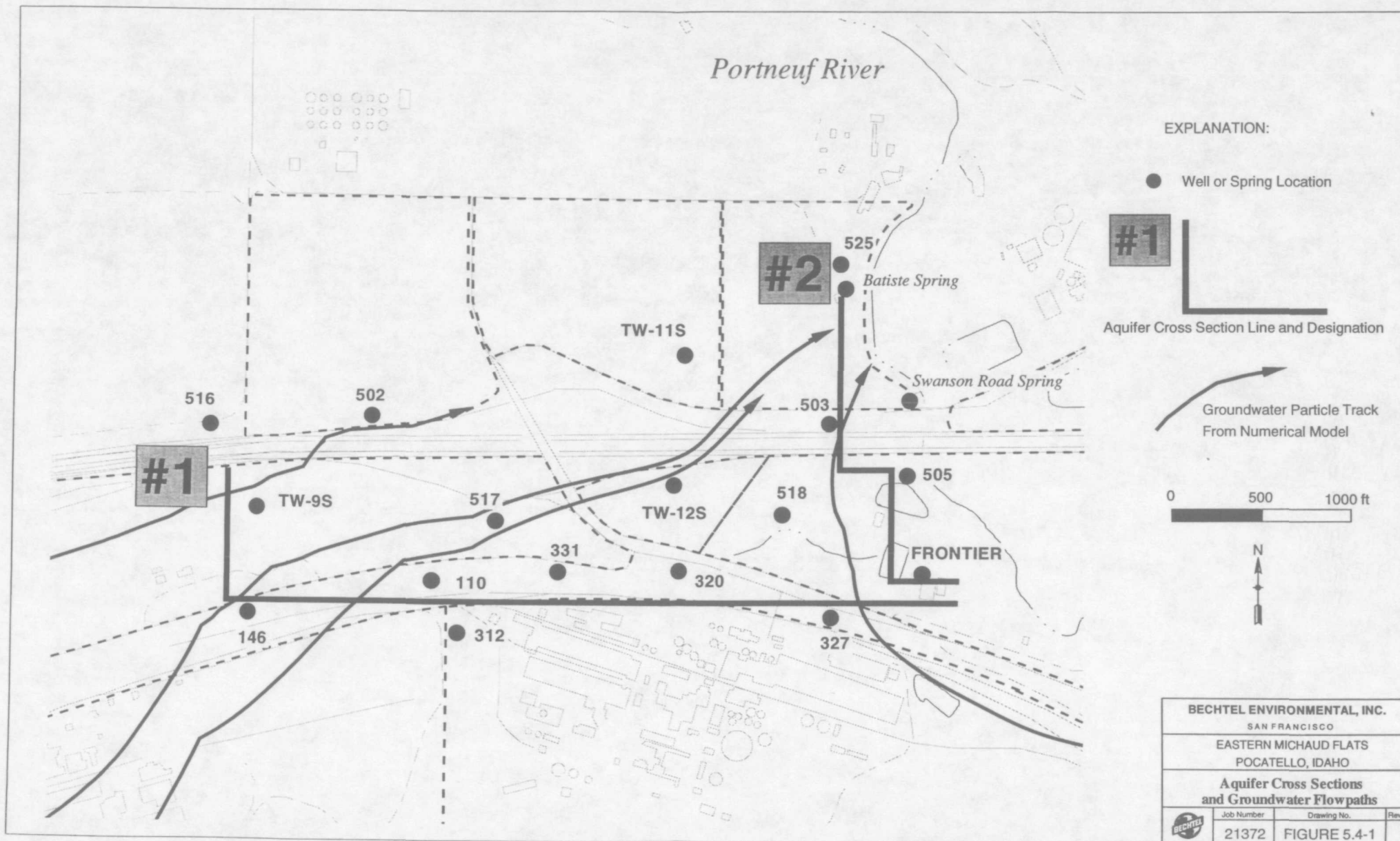
CONSTITUENT	UPSTREAM (mg/l)	EMF-RELATED (mg/l)	BACKGROUND, STP, AND OTHER SOURCES (mg/l)	DOWNSTREAM (mg/l)
Nitrate	0.60	0.0325	1.44	2.07
Sulfate	43	3.0	14	60
Total Phosphorus	0.10	0.06	0.417	0.59

TABLE 5.4-7
AVERAGE LOW FLOW SOLUTE CONCENTRATIONS IN PORTNEUF RIVER

CONSTITUENT	AVERAGE UPSTREAM CONCENTRATION (mg/l)		AVERAGE DOWNSTREAM CONCENTRATION (mg/l)	
	JULY 1992	1992-1993	JULY 1992	1992-1993
Nitrate	0.087	0.60	2.07	2.07
Sulfate	39	43	59	60
Total Phosphorus	0.037	0.010	0.50	0.59

Notes: Upstream flow rates measured during the July 1992 sampling event were 18.4 cfs at station 25 and 20 cfs at station 16, which are very low flow conditions.

Surface and Subsurface Characterizations
Figures for Section 5.4



Solute Fluxes (kg/min)

Sulfate	2.7	2.7
Nitrate	0.055	0.028
Arsenic	0.00029	0.00020
Selenium	0.00015	0.000032
Total P	0.065	0.060

Portneuf River

EXPLANATION:

#1

Aquifer Cross Section Line and Designation

#2

Batiste Spring

Swanson Road Spring

#1

0 500 1000 ft

N

BECHTEL ENVIRONMENTAL, INC.
SAN FRANCISCO

EASTERN MICHAUD FLATS
POCATELLO, IDAHO

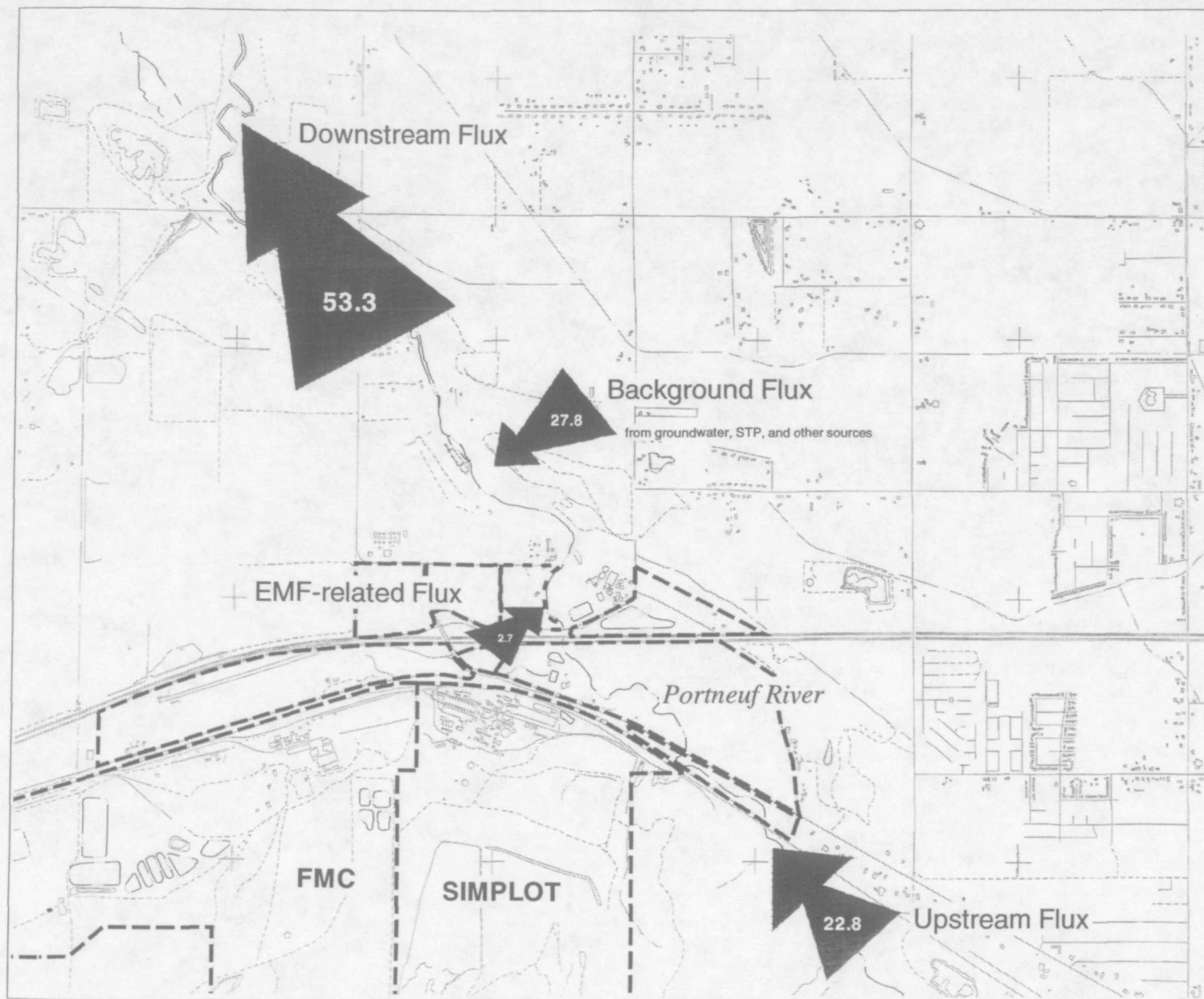
Solute Transport in Shallow Aquifer



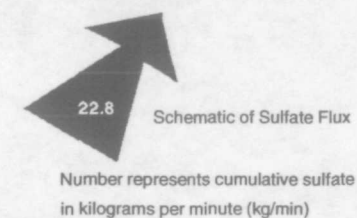
Job Number
21372

Drawing No.
FIGURE 5.4-2

Rev.



EXPLANATION:



--- EMF Site Boundary



SCALE
2500 ft

BECHTEL ENVIRONMENTAL, INC.

SAN FRANCISCO

EASTERN MICHAUD FLATS

POCATELLO, IDAHO

Regional Sulfate Fluxes



Job Number

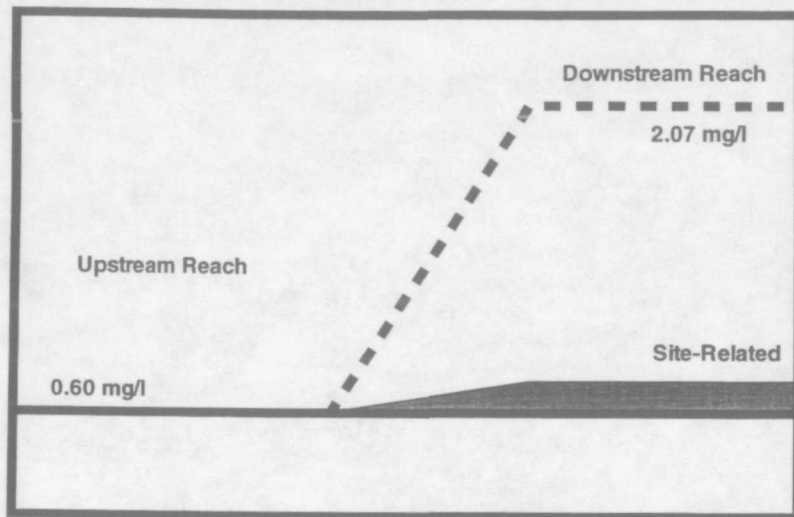
21372

Drawing No.

FIGURE 5.4-3

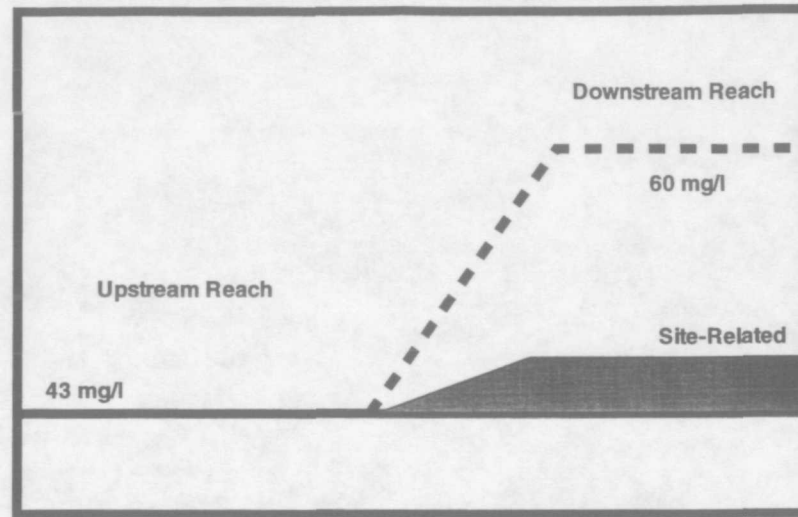
Rev.

Nitrate Concentration (mg/l)



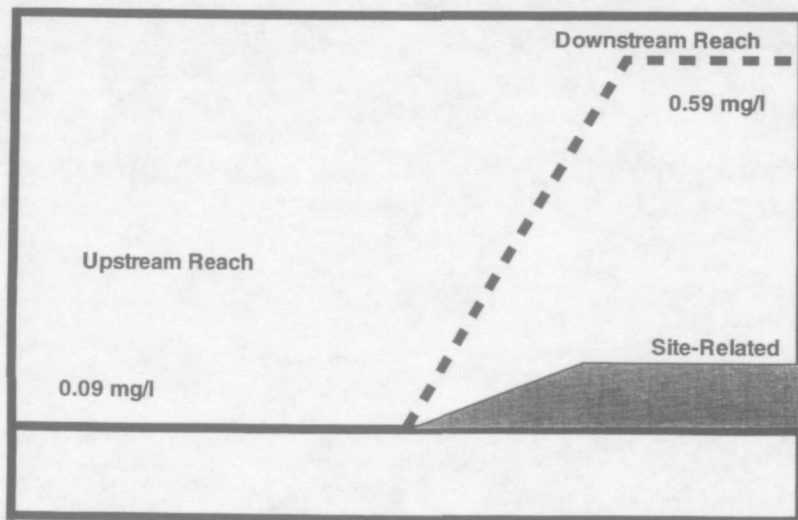
Distance Along Portneuf River (not to scale)

Sulfate Concentration (mg/l)



Distance Along Portneuf River (not to scale)

Total Phosphorus Concentration (mg/l)



Distance Along Portneuf River (not to scale)

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Concentration Profiles Along
Portneuf River



Job Number
21372

Drawing No.
FIGURE 5.4-4

Rev

5.5 SUMMARY

EMF RI data indicate that the primary pathways through which constituents originating from the EMF facilities are transported are air, the vadose zone, and groundwater. (The air pathway is addressed in RI Report, Part III.) The data further indicate that the EMF-related constituents transported via the vadose zone and groundwater pathways originate at source areas to which a sustained artificial head, containing high solute concentrations, is or was applied. These source areas include FMC's former unlined ponds and Simplot's gypsum stack.

The principal factors affecting migration in the vadose zone are seepage rates and seepage fluid and soil chemistry. Seepage from the EMF facilities has been significantly reduced over the last several years due to changes in facility operations (e.g., pond closures, gypsum stack operational changes). In addition to reductions in seepage, the presence of alkaline, carbonate-rich loess beneath large portions of the EMF facilities has contributed to the attenuation of some metals in the vadose zone. The generally low-pH seepage reacts with the loess, resulting in an increase in the pH of the fluid, thereby inducing metals precipitation.

The principal factors affecting migration in the saturated zone are aquifer geochemistry and advective mixing. When seepage reaches the saturated zone, it reacts with the neutral-to-basic pH groundwater, resulting in precipitation of additional constituents that were present in the migrating fluids. In addition, the introduction of fluoride to groundwater is limited because the groundwater is naturally at the saturation point of fluoride. Sulfate concentrations are also limited by aquifer geochemistry.

The most significant factor in the reduction of groundwater constituent concentrations is advective mixing. Mixing of small volumes of EMF-affected groundwater with large volumes of unaffected groundwater within the EMF aquifer system substantially reduces the concentration of all constituents, including conservative, non-attenuating solutes such as sulfate, along the groundwater flowpath. The influence of advective mixing has been illustrated with a groundwater solute flux analysis, which shows that the mass of EMF-related constituents

discharged to surface water is small compared with background contributions. In addition, it confirmed that EMF fluxes have no measurable influence on surface water chemistry.